UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Southeast Regional Office
263 13th Avenue South
St. Petersburg, Florida 33701-5505
http://sero.nmfs.noaa.gov

F/SER31: KL
SER-2012-03723

MAR 07 2014

Mr. Eric Summa
Jacksonville District Corps of Engineers
Department of the Army
P.O. Box 4970
Jacksonville, Florida 32232

Mr. Chris McArthur
United States Environmental Protection Agency
Region 4
Atlanta Federal Center
61 Forsyth Street
Atlanta, Georgia 30303

Re: Port Everglades Expansion Project, Broward County, Florida

Dear Mr. Summa and Mr. McArthur:

The enclosed document constitutes the National Marine Fisheries Service's (NMFS) Biological Opinion based on our review of the U.S. Army Corps of Engineers' (USACE) planned dredging activities for the expansion of Port Everglades, and the Environmental Protection Agency's (EPA) expansion of the Offshore Dredged Material Disposal Site as an interrelated and interdependent activity. This Opinion is based on project-specific information provided in the consultation packages in addition to NMFS's review of published literature. This Opinion analyzes the project effects on whales, Johnson's seagrass, sea turtles, smalltooth sawfish, staghorn coral and six corals proposed for listing, as well as designated critical habitat for elkhorn and staghorn coral and proposed critical habitat for the NWA DPS of loggerhead sea turtles. We believe that the proposed project is likely to adversely affect, but is not likely to jeopardize, the continued existence of sea turtles, Johnson's seagrass, staghorn coral and proposed corals, and is not likely to destroy or adversely modify designated critical habitat for elkhorn and staghorn corals.

This Opinion includes a conference opinion on 6 species of proposed corals, and the proposed reclassification of staghorn coral from threatened to endangered. As such, if the proposed listings and reclassification are finalized in June 2014, the USACE will need to contact NMFS to determine the mechanism for authorizing the take of corals necessary to implement this action as proposed. Since the USACE has requested conference consultation on the proposed species, at the proper time they must also request that this Conference Opinion be confirmed as NMFS's Biological Opinion.

We look forward to further cooperation with you on other USACE and EPA projects to ensure the conservation and recovery of our threatened and endangered marine species. If you have any questions regarding this consultation, please contact Kelly Logan by phone at 727-460-9258 or by email at Kel.Logan@noaa.gov.

Sincerely,

Miles M. Croom

for Roy E. Crabtree, Ph.D.
Regional Administrator

Enclosure
File: 1514-22.F.4

Endangered Species Act - Section 7 Consultation
Biological Opinion

Agency:	United States Army Corps of Engineers (USACE) and United States Environmental Protection Agency (EPA)
Applicant:	USACE Planning and Civil Works Division
Activity:	Dredging and Expansion of Port Everglades, Broward County, Florida
Consulting Agency:	National Marine Fisheries Service (NMFS) Southeast Regional Office Protected Resources Division
	NMFS Consultation No. SER-2012-03723
Date Issued:	MAR 07 2014
Approved By:	Miles M. Croom for Roy E. Crabtree, Ph.D. Regional Administrator

Glossary of Commonly Used Acronyms

CCA	Crustose Coralline Algae
DPS	Distinct Population Segment
DWH	Deepwater Horizon
EPA	Environmental Protection Agency
ESA	Endangered Species Act of 1973
HCD	Habitat Conservation Division
ITS	Incidental Take Statement
MMPA	Marine Mammal Protection Act of 1972
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
ODMDS	Ocean Dredged Material Disposal Site
RBO	Regional Biological Opinion
RPMs	Reasonable and Prudent Measures
SARBO	South Atlantic Regional Biological Opinion
SEFSC	Southeast Fisheries Science Center
SERO	Southeast Regional Office
STSSN	Sea Turtle Stranding and Salvage Network
USACE	United States Army Corps of Engineers
USFWS	United States Fish and Wildlife Service

Background

Section 7(a)(2) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. §1531 et seq.), requires that each federal agency ensure that any action authorized, funded, or carried out by the agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of those species. When the action of a federal agency may affect a protected species or its critical habitat, that agency is required to consult with either National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the protected species that may be affected.

Consultations on most listed marine species and their designated critical habitat are conducted between the action agency and NMFS. Consultations are concluded after NMFS determines the action is not likely to adversely affect listed species or critical habitat or issues a biological opinion ("opinion") that determines whether a proposed action is likely to jeopardize the continued existence of a federally-listed species, or destroy or adversely modify federally-designated critical habitat. The opinion also states the amount or extent of listed species incidental take that may occur and develops nondiscretionary measures that the action agency must take to reduce the effects of said anticipated/authorized take. The opinion may also recommend discretionary conservation measures. No incidental destruction or adverse modification of critical habitat may be authorized. The issuance of an opinion detailing NMFS's findings concludes ESA Section 7 consultation.

This document represents NMFS's Biological Opinion ("Opinion") based on our review of impacts associated with the USACE's proposed dredging and expansion of Port Everglades ("Port Everglades Expansion project"), and the U.S. Environmental Protection Agency's (EPA) proposed expansion of the Ocean Dredged Material Disposal Site (ODMDS). This Opinion analyzes project effects on Johnson's seagrass, sea turtles, whales, smalltooth sawfish, staghorn coral, and designated critical habitat for elkhorn and staghorn corals in accordance with Section 7 of the ESA. NMFS based this Opinion on project information provided by the USACE as well as published literature and the best available scientific and commercial information. It is NMFS's Biological Opinion that the action, as proposed, is not likely to jeopardize the continued existence of Johnson's seagrass, sea turtles or staghorn coral, and is not likely to destroy or adversely modify the designated critical habitat for elkhorn and staghorn corals.

NMFS has also proposed to list 6 additional coral species, which may be found within the action area. The 6 species are: *Orbicella annularis, Orbicella faveolata, Orbicella franksi, Mycetophyllia ferox, Dichocoenia stokesii,* and *Agaricia lamarcki*. In addition, NMFS has proposed to reclassify *Acropora cervicornis* and *Acropora palmata* from threatened to endangered (77 FR 73220, December 7, 2012). The USACE has requested a formal Conference Opinion for these proposed corals. A conference consultation on the potential effects of the action on proposed critical habitat for the Northwest Atlantic DPS of loggerhead sea turtles is also included. Conference is a process of early interagency cooperation involving informal or formal discussions between the action agency and NMFS pursuant to Section 7(a)(4) of the ESA regarding the likely impact of an action on proposed species or proposed critical habitat. Conferences are: (1) required for proposed federal actions likely to jeopardize proposed species, or destroy or adversely modify proposed critical habitat; (2) designed to help federal agencies identify and resolve potential conflicts between an action and species conservation early in a project's planning; and (3) designed to develop recommendations to minimize or avoid adverse effects to proposed species or proposed critical habitat. [50 CFR §402.02, 50CFR §402.10]. This document will incorporate NMFS's Conference Opinion for the 6 proposed coral species and proposed loggerhead critical habitat based on our review of impacts associated with the Port Everglades Expansion project and expansion of the ODMDS. It is NMFS's Opinion that the proposed action will not jeopardize the continued existence of any of these proposed species or destroy or adversely modify the proposed critical habitat for the loggerhead Northwest Atlantic DPS.

BIOLOGICAL OPINION

1 Consultation History

On March 25, 2002, the USACE submitted a biological assessment for the Port Everglades Expansion project. Due to changes in the project design, listing of new species and designation of new critical habitat, ongoing information requests, and workload, consultation could only be completed recently. The following is a list of important consultation dates and activities:

- 2003 – Changes in ship simulations resulted in potential changes to project impacts; the USACE requested the consultation be suspended.
- September 17, 2004 – USACE submitted a revised biological assessment.
- May 9, 2005 – NMFS proposed to list elkhorn and staghorn corals.
- Late May 2005 – NMFS and USACE discussed the survey information previously provided to NMFS and determined that no additional surveys would be completed at that time.
- June 23, 2005 – USACE determined that the Port Everglades Expansion project may affect, but is not likely to adversely affect, elkhorn and staghorn coral.
- March 29, 2006 – USACE provided additional details and graphics regarding the disposal areas.
- May 9, 2006 – NMFS listed elkhorn and staghorn corals as threatened under the ESA.
- May 2006 – USACE conducted a reef survey to provide additional details on species composition at the end of the entrance channel.
- June 21, 2006 – USACE and NMFS biologists met to discuss project status and transfer information.
- June 23, 2006 – USACE sent draft of Port Everglades Reef Report.
- July 25, 2006 – USACE and NMFS met to discuss results of Port Everglades Reef Report.
- August 11, 2006 – NMFS provided comments on Reef Report to USACE. NMFS recommended that USACE complete a survey designed specifically to identify the presence and abundance of elkhorn and staghorn corals.
- August 18, 2006 – NMFS sent letter to USACE stating our belief that the coral reef survey study design was flawed.
- August-September, 2006 – USACE and NMFS coordinated by emails and agreed to a revised project area and updated seagrass reports.
- October 18, 2006 – USACE provided a letter responding to NMFS's determination that the original coral resource survey design was flawed.
- March 26, 2008 – NMFS sent a letter to USACE stating our concern that *Acropora cervicornis* may occur closer than the stated 3,500 feet (ft) from the entrance channel.
- April 28, 2008 – USACE and NMFS met to discuss project timeline and coral survey methodology. NMFS and USACE agreed to develop alternative survey methods for navigational channels in order to provide for human safety.
- December 2009 – Dial Cordy, Inc. finalized the Benthic and Fish Community Assessment Report.
- Summer 2010 – USACE conducted a new *Acropora* survey using the new alternative methods.

- October 2010 – USACE submitted the *Acropora* Survey Final Report to NMFS.
- October 12, 2010 – NMFS and USACE reviewed video and results of *Acropora* survey.
- August 2, 2011 – NMFS received information on a new *Acropora* survey conducted by the U.S. Navy. The survey found colonies of *Acropora* on the outer reef near the Port Everglades entrance channel. NMFS requested a hold on consultation until completion of the Navy report.
- December 2011 – NMFS received a copy of the final report from the Navy.
- February 13, 2012 – NMFS provided a copy of the final report from the Navy to the USACE.
- May 1, 2012 – NMFS and USACE met to discuss ongoing projects and timelines. During the meeting, NMFS requested that the USACE submit a complete consultation package for the Port Everglades Expansion project.
- September 5, 2012 – USACE submitted the final consultation package to NMFS.
- September 27, 2012 – USACE and NMFS met in St. Petersburg to discuss project status, possible alternatives, and ongoing NMFS concerns.
- October 16, 2012 – NMFS and USACE met in West Palm Beach with Dial Cordy, Inc. to discuss the towed video *Acropora* survey footage.
- December 7, 2012 – NMFS proposed to list 7 additional corals in the greater Caribbean region, 6 of which are documented in the project area.
- January 2, 2013 – USACE requested initiation of a formal conference opinion on the 6 proposed corals within the project area.
- January-May 2013 – NMFS worked on development of alternative coral reef mitigation options.
- May-October 2013 – NMFS and USACE developed "blended mitigation alternatives" to address impacts to coral reef resources.
- November 19-20, 2013 – NMFS and USACE met to resolve differences and composed a framework for a blended mitigation approach.

During the meeting held in St. Petersburg, Florida on November 19 and 20, 2013, NMFS and USACE resolved many of the remaining differences and completed the framework for a blended mitigation plan. USACE provided NMFS with a revised project description via email dated November 22, 2013 and we initiated formal consultation on that date.

2 Description of the Proposed Action

This consultation addresses the expansion of Port Everglades located within Hollywood, Broward County, Florida (see Figure 1). The total time frame for dredging is approximately 5 years. The propagation and outplanting of corals, a mitigation component of the proposed action, is expected to take 7 years with monitoring continued for an additional 3 years. The proposed project components are as follows (see Figure 2):

1. Deepen, widen, and extend the Outer Entrance Channel from an existing 45-ft project depth over a 500-ft channel width to 57 ft deep by 800 ft wide and extend it 2,200 ft seaward

2. Deepen the Inner Entrance Channel from 42 ft to 50 ft
3. Deepen the Main Turning Basin from 42 ft to 50 ft
4. Widen the rectangular shoal region to the southeast of the Main Turning Basin (Widener) by approximately 300 ft and deepen to 50 ft
5. Widen the Southport Access Channel in the proximity of Berths 23-26, referred to as "The Knuckle," by about 250 ft and relocate the U.S. Coast Guard (USCG) facility farther east on USCG property
6. Shift the existing 400-foot-wide Southport Access Channel about 65 ft to the east from approximately Berth 26 to the south end of Berth 29 to provide a transition back to the existing federal channel limits
7. Deepen the Southport Access Channel from about Berth 23 to the south end of Berth 32 from 42 ft to 50 ft
8. Deepen the Turning Notch, including the expanded portion, from 42 ft to 50 ft with an additional 100-foot north-south widening parallel to the Southport Access Channel on the eastern edge over a length of about 1,845 ft, and widen the western edge of the channel approximately 130 ft to provide access to the Turning Notch from the existing federal channel
9. Pre-treat rock substrates as necessary, including blasting
10. Dispose of dredged material not used for mitigation construction at the ODMDS, located east of the Port
11. Create approximately 5 acres of boulder reef
12. Relocate approximately 11,500 corals from within the impact area to the artificial boulder reef
13. Propagate and outplant corals, including between 35,000 and 50,000 *Acropora cervicornis* colonies
14. Temporarily relocate existing Aids to Navigation (ATONs) adjacent to the channel

All dredge depths may include up to 2 ft overdredge, meaning that the contractor(s) will be able to dredge to 2 ft below all depths identified above. Exact dredging methods will be determined later and will be dependent upon to whom the USACE awards the contract. Hopper dredges may be used prior to beginning the expansion to remove accumulated shoal material from the existing channel. Sand, silt, clay, soft rock, rock fragments, and loose rock will be removed via clamshell or suction dredge. Where contractors encounter hard rock, the USACE anticipates that explosives, and/or large cutterhead equipment will be used to remove the rock. Approximately 5.47 million cubic yards of material will be removed from all dredging activities to complete the expansion.

The use of explosives will be limited to areas inshore of the outer reef. The USACE estimates that up to 50% of the area to be dredged may require pre-treatment of hard substrate, and that there may be up to 900 days on which blasting takes place over the course of the 5-year construction period. The USACE will require the contractor(s) to use the following conservation measures to protect marine mammals and sea turtles:

1. A danger zone will be determined based on the explosive weight used and its effects during an open water detonation. This will give a conservative danger zone because the USACE will only use confined blasting techniques.

2. A combination of aerial observers, on water observers, and observers on the drill vessel will monitor the danger zone.

3. Any marine mammal or sea turtle within the danger zone shall not be forced to move out of these zones. Detonation shall not occur until the animal has moved out of the danger zone of its own volition.

4. In the event a protected species is injured or killed during the use of explosives, the USACE will immediately notify NMFS and engage in additional consultation prior to further use of explosives.

5. If explosives are used, the USACE will place the explosives in strategically oriented pre-drilled holes. These holes will be stemmed with angled gravel to direct the explosive energy into the rock.

6. The weight of explosives to be used in each blast will be limited to the lowest poundage of explosives that can adequately break the rock.

7. Drill patterns are restricted to a minimum of 8-foot separation from a loaded hole.

8. Hours of blasting are restricted from 2 hours after sunrise to 1 hour before sunset to allow for adequate observation of the project area for protected species.

9. Selection of explosive products and their practical application method must address vibration and air blast (overpressure) control for protection of existing structures and marine wildlife.

10. Loaded blast holes will be individually delayed to reduce the maximum pounds per delay at point detonation, which in turn will reduce the mortality radius.

11. The blast design will match the energy in the "work effort" of the borehole to the rock mass or target for minimizing excess energy vented into the water column or hydraulic shock.

12. Delay timing adjustments to a minimum of 8 milliseconds (ms) between delay detonations to stagger the blast pressures and prevent cumulative addition of pressures in the water.

13. Due to the likelihood of a large number of manatees in the area during the winter months, USACE has agreed as part of the ESA consultation with USFWS not to blast between November 15 and March 15. This will also help protect whales which migrate through this area in the early spring and late fall.

14. Test blasts will be performed prior to the actual project blasting. Observers will also be stationed to observe for endangered species prior to test and project blasts.

Safety radii are as follows:

1. The Danger Zone (NMFS refers to this as the Caution Zone): The radius in feet from the detonation beyond which no expected mortality or injury from an open water explosion is likely to occur (NMFS 2005c). The danger zone (ft) = 260 [79.25 m] X the cube root of weight of explosives in lbs per delay (equivalent weight of TNT).

2. The Safety Zone is the approximate distance in feet beyond which injury (Level A harassment as defined in the MMPA) is unlikely to occur from an open water explosion (NMFS 2005c). The safety zone (ft) = 520 [158.50 m] X cube root of weight of explosives in lbs per delay (equivalent weight of TNT).

3. The Watch Zone is 3 times the radius of the Danger Zone to ensure that animals entering or near the Exclusion Zone are spotted and appropriate actions can be implemented before or as they enter any impact areas (i.e., a delay in blasting activities).

4. The Exclusion Zone extends to 500 ft outside the Danger Zone radius. Detonation will not occur if a marine mammal or sea turtle may be within that zone (based on observational data).

Monitoring/watch plan. A watch plan will be formulated based on the required monitoring radii and optimal observation locations. The watch plan will be consistent with the program that was utilized successfully at Miami Harbor in 2005 and will consist of at least 5 observers including at least 1 aerial observer, 2 boat-based observers, and 2 observers stationed on the drill barge. A 6th observer will be placed in the most optimal observation location (boat, barge, fixed structure, shore, or aircraft) on a day-by-day basis depending on the location of the blast and the placement of dredging equipment, as determined by the blaster in charge and the chief protected species observer. This process will ensure complete coverage of the 3 zones as well as any critical areas. The watch will begin at least 1 hour prior to each blast and continue for one-half hour after each blast (Jordan et al. 2007).

Studies have shown that stemmed blasts have up to a 60% to 90% decrease in the strength of the pressure wave released, compared to open-water blasts of the same charge weight (Hempen et al. 2007; Hempen et al. 2005; Nedwell and Thandavamoorthy 1992). However, unlike open-water blasts, very little documentation exists on the effects that confined blasting can have on marine animals near the blast (Keevin et al. 1999). The blast mitigation procedures detailed above, in particular the rigorous observer program, have been successfully used in several USACE projects (i.e., San Juan Harbor, Puerto Rico, in 1994; Miami Harbor in 2005; and Wilmington Harbor in 2012).

The USACE will require the contractor(s) to follow the Terms and Conditions in NMFS's 1997 Regional Biological Opinion (RBO) on Hopper Dredging along the South Atlantic Coast. The

1997 RBO incorporates (by reference) NMFS's 1995 Biological Opinion on hopper dredging of channels and beach nourishment activities in the southeastern United States from North Carolina through Florida East Coast. The contractor(s) will be required to follow the Terms and Conditions in the 1997 and 1995 Biological Opinions mentioned above, with the exception of the conditions related to the southeast United States' North Atlantic Right Whale calving area, because the proposed project is not located in or near the calving area. The USACE will also require the contractor(s) to follow the enclosed NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions*, dated March 23, 2006 (Appendix B).

Based on info from the DEIS (USACE 2013), current impact estimates show expanding the Port's Outer Entrance Channel will directly affect (via dredging) less than 8 acres of seagrasses, including 4.67 acres of Johnson's seagrass, 1.2 acres of mangroves, and 21.66 acres of coral reef habitat. The USACE will use mitigation credits at Westlake Park for losses of seagrass and mangroves. Further, approximately 98 to 118 acres of coral reef habitat may be impacted by sedimentation and anchoring of dredging vessels, depending on the methods ultimately used. The USACE anticipates that the methods used will likely result in the lower end of these effects; thus, the Port Everglades Expansion project includes mitigation actions to compensate for impacts to approximately 120 acres containing corals and coral reef. NMFS and the USACE worked together to create a "blended" mititgation plan, consisting of artificial reef creation and enhancement, and propogation and transplantion of sponges and corals, including listed staghorn corals, to natural reefs. The draft Habitat Equivalency Analysis, draft mitigation plan, and final meeting notes from collaborative meetings held in November 2013 (all of which can be found within the consultation documents for this project) were used to develop the following agreed-upon elements of mitigation:

1. *Creation of approximately 5 acres of artificial boulder reefs.*
   USACE proposes to deploy piles of limestone that have either been quarried and transported to the mitigation area, or dredged from the channel construction areas. The exact layout and artificial reef sites will be determined as part of the final mitigation plan.

2. *Relocation of approximately 11,500 corals from the impact area to artificial boulder reefs.*
   Approximatley 11,500 corals are proposed to be relocated from the impact area to the created artificial boulder reefs. Corals are removed from their natural substrate using hand tools, such as chisels and hammers. Depending on the distance to the outplanting site, the dislocated corals are transported either underwater by the diver or in seawater-filled containers at the surface. The corals are then attached to the artificial boulder reef using technologically proven standards. The density of the outplants will approximate the density in which they occurred in their natural state.

3. *Propagation and outplantation of corals and sponges, including between 35,000 and 50,000 staghorn coral colonies*
   The USACE proposes to enhance partially degraded reef sites near to, but not directly in or adjacent to, the area impacted by the Port Everglades Expansion project. This proposed reef mitigation project would enhance degraded reefs through the placement (outplanting) of regionally appropriate corals and sponges at appropriate density and

numbers as determined by NMFS. The organisms for outplanting would be sourced from corals and sponges of opportunity or propagated within in situ or ex situ coral nurseries. The exact numbers and proportions of the various coral species outplanted will be determined as part of the final mitigation plan. The USACE anticipates that between 35,000 and 50,000 colonies of staghorn coral will be part of the suite of outplanted organisms. The USACE anticipates that this portion of the mitigation (setting up or augmenting nurseries, growing corals and outplanting enough to meet the mitigation goals) will take up to 7 years to meet their mitigation requirements,with monitoring continuing for up to an additional 3 years.

Coral nurseries currently exist within Broward County and could be the source of the outplanted corals. However, it is possible that the volume of corals needed for the Port Everglades mitigation actions will exceed the capacity of existing nurseries; thus, new nurseries may be established. Scientifically vetted best practices for nursery propagation, outplanting, and monitoring have been developed and used by nursery managers in the Florida Keys, Broward County, Puerto Rico, U.S. Virgin Islands, and other Caribbean islands to reproduce *Acropora* spp. asexually (e.g., Johnson et al. 2011). Typical coral nursery establishment includes collection of small fragments less than 5 centimeters in diameter from the reef and holding the fragments in an underwater or tank-based nursery environment through their juvenile life stage. Corals of opportunity (i.e., fragments or colonies found already broken or dislodged from reef substrates) are the preferred source of fragment stock for the nursery; however, sometimes wild colonies must be sampled to obtain nursery stock. Small branch clippings from wild donor colonies can be collected using a variety of cutting tools, including stainless steel surgical bonecutters, diagonal electrical wire cutters, needle-nose pliers, etc. For colonies with thicker branches, PVC cutters have also proven effective. Branches are cut cleanly and evenly to ensure optimal survival of the fragment as well as rapid healing and recovery of the donor colony (Johnson et al. 2011). Offshore nurseries are sited to balance a number of factors including, among others, appropriate habitat and water quality conditions, potential for future impacts, and permitting. The physical and genetic origin of each coral is tracked from fragment collection to ensure that both nursery and outplanting operations are done in a scientifically responsible way. Regular maintenance is performed on nursery structures and the corals themselves to ensure all are free of coral competitors and predators. Once coral fragments have grown to a size where the probability of survival on natural reefs has increased to an acceptable level (this usually requires 12 to 18 months), the corals are outplanted to the natural reef.

Similar to nursery siting, outplanting sites are selected balancing several factors to maximize success. During outplanting, care is taken to ensure external stresses are minimized and that a population with an acceptable level of genetic diversity and environmental tolerance is developed. Algae and predators are periodically removed from the outplanted corals until they are firmly established on the reef. A stock population is maintained within the nursery to provide new colonies for outplanting. Corals can be attached directly to the reef or using attachment platforms like masonry nails or cement pucks. Outplanted corals can be wedged into holes or crevices, or secured using epoxy, cement, wire, or plastic ties.

In addition to the above activities for Port expansion, NMFS received a request for consultation dated August 26, 2013, from EPA to expand the ODMDS located offshore of Fort Lauderdale, Broward County, Florida (see Figure 3). NMFS has determined that this project is interrelated to and interdependent with the Port Evergaldes Expansion project, therefore it will be included in this Biological Opinion. The proposed project includes expanding the ODMDS from the existing 0.9 square nautical miles (nm^2) to an area of 3.21 nm^2, which will have a north-south oriented release zone. The Port Everglades Expansion project includes disposal of up to 5.47 million cubic yards (mcy) of dredged material within the ODMDS. The western edge of the site is located 3.3 nm offshore and the center of the site is located approximately 4 nm offshore. Water depths range from 604 to 735 ft. Previously collected sidescan sonar data (EPA 2004) and data collected from the OSV (Ocean Survey Vessel) Bold's site designation survey in May 2011 (ANAMAR 2012) indicate the bottom within the expansion area is primarily a homogenous mix of sand, silt, and clay with scattered rubble. There are approximately 12.85 acres of hardbottom within the expansion area; however, it is located below the 30-meter depth contour and is not considered critical habitat for any listed coral species, nor does it function as refuge habitat for sea turtles.

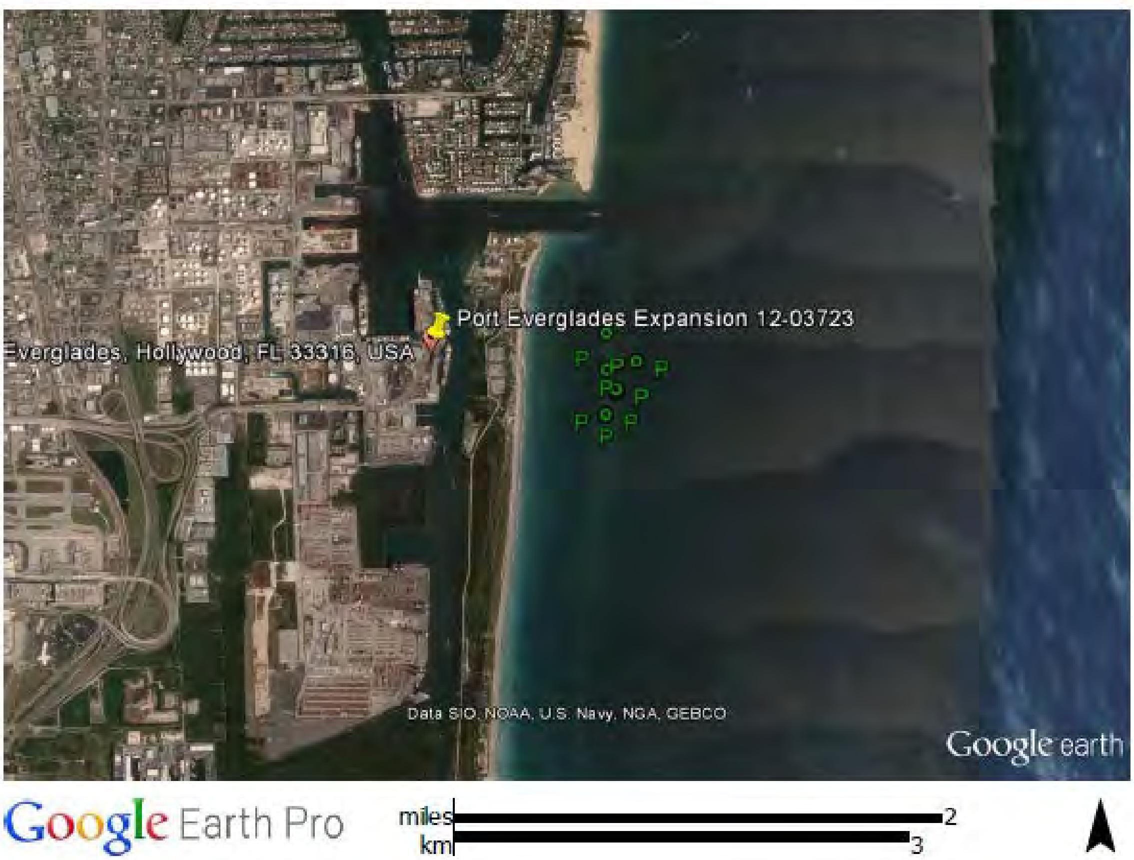

Figure 1. Location of Port Everglades Expansion project. Known colonies of *Acropora* corals are indicated in green.

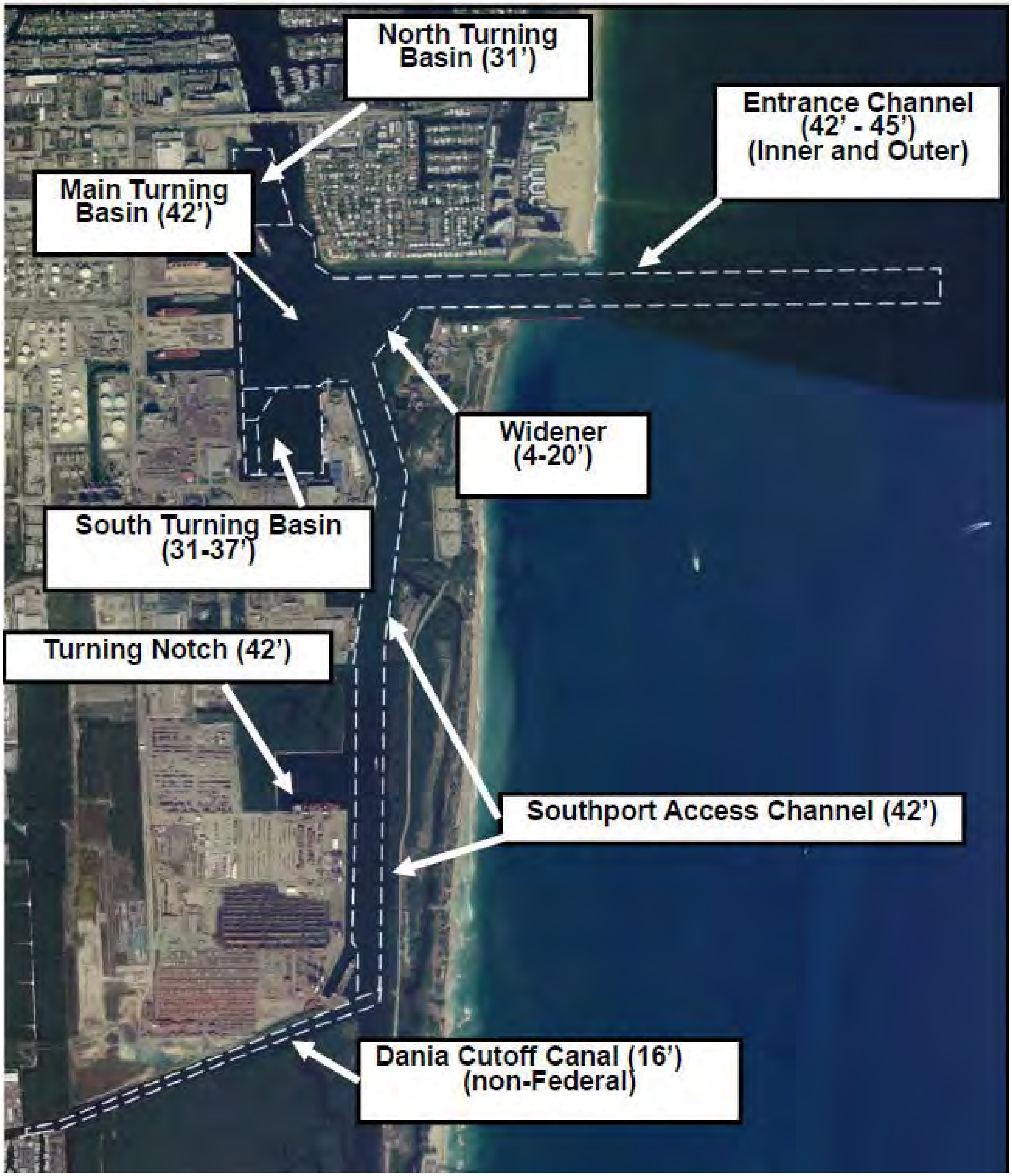

Figure 2. Port Everglades Expansion project components showing current depths. The Dania Cutoff Canal and the North Turning Basin are no longer part of the project (figure courtesy of USACE).

Figure 3. Location of EPA's ODMDS expansion project

3 Action Area

The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02). The action area for this project includes the Port Everglades Harbor, which is located in Hollywood, Broward County, Florida. Port Everglades includes several areas within the Intracoastal Waterway and the access channel, which extends into the Atlantic Ocean. The action area also includes a 150-ft buffer zone surrounding the access channel, in which effects of dredging will likely occur. The action area also includes the spoil disposal sites, which consist of a site in the nearshore Atlantic Ocean off Broward County where the boulder mitigation reef will be created, the ODMDS in the Atlantic Ocean off Broward County, and the routes of vessel travel to and from the disposal sites. Last, the action area includes the coral nursery and outplanting sites within the Florida Reef Tract offshore.

4 Status of Listed Species and Critical Habitat

The following endangered (E), threatened (T), and proposed species (P), and designated critical habitat under the jurisdiction of NMFS may occur in or near the action area.

Table 1. Listed and Proposed Species and Critical Habitat Likely to Occur in or Near the Project Area

Common Name	**Listed Species** Scientific Name	Status
Turtles		
Green sea turtle	*Chelonia mydas*[1]	E/T
Kemp's ridley sea turtle	*Lepidochelys kempii*	E
Leatherback sea turtle	*Dermochelys coriacea*	E
Loggerhead sea turtle	*Caretta caretta*[2]	T
Hawksbill sea turtle	*Eretmochelys imbricata*	E
Fish		
Smalltooth sawfish	*Pristis pectinata*[3]	E
Invertebrates and Marine Plants		
Staghorn coral	*Acropora cervicornis*	T; P-E[4]
Johnson's seagrass	*Halophila johnsonii*	T
Marine Mammals		
Humpback whale	*Megaptera novaeangliae*	E
Sperm whale	*Physeter macrocephalus*	E
Designated Critical Habitat		
Elkhorn/staghorn coral		
Common Name	**Proposed Species** Scientific Name	Status
Invertebrates		
Elliptical star coral	*Dichocoenia stokesii*	P-T[5]
Lamarck's sheet coral	*Agaricia lamarcki*	P-T
Rough cactus coral	*Mycetophyllia ferox*	P-E
Lobed star coral	*Orbicella annularis*	P-E
Mountainous star coral	*Orbicella faveolata*	P-E

[1] Green turtles are listed as threatened except for the Florida and Pacific coast of Mexico breeding populations, which are listed as endangered.
[2] Northwest Atlantic Ocean (NWA) DPS.
[3] The U.S. DPS.
[4] *Acropora cervicornis* is currently listed as threatened and proposed for reclassification to endangered on December 7, 2012.
[5] All proposed corals were listed in the Federal Register on December 7, 2012.

Knobby star coral	*Orbicella franksi*	P-E
Proposed Critical Habitat		
Loggerhead sea turtle	Migratory and Breeding Habitat	Within Critical Habitat Unit Logg-N-19

4.1 Species and Critical Habitat Not Likely to be Adversely Affected

NMFS has analyzed the routes of potential project effects in the marine environment on 5 species of sea turtles (loggerhead, Kemp's ridley, leatherback, hawksbill, and green), smalltooth sawfish, humpback whales, and sperm whales from the proposed action. We have determined the potential routes of effects to sea turtles and smalltooth sawfish include (1) injury or death from potential interactions with and operation of dredges and blasting, and (2) avoidance of the area during construction operations due to disturbance caused by blasting, dredging, and placement of dredged materials in the various disposal sites (ODMDS, and the artificial reef mitigation site). Loss of foraging habitat within the dredge footprint could also affect sea turtles. The potential routes of effects to whales include injury or death from potential interactions with hopper dredges during dredging and disposal of dredged material in the ODMDS, injury or death from potential blasting, and temporary avoidance of areas during construction. Of these, only interactions with hopper dredges have the potential for adverse effects, and only for certain turtle species, as discussed below and in the Effects of the Action section.

Smalltooth Sawfish

Smalltooth sawfish are unlikely to be found within the existing channel area but may be found in the mangrove areas located at the western edge of John U. Lloyd State Park, although no sawfish have been reported. In the unlikely event a sawfish is present in the project area, sawfish should not be injured or killed by the dredging or construction activities because the dredges advance relatively slowly (the cutterhead dredges and mechanical-type dredges that are feasible to use in these areas are very slow, almost stationary) and are noisy, giving mobile sawfish the opportunity to get out of the way. Due to the sawfish's mobility, ability to detect the approaching draghead, and apparent avoidance behavior, the risk of injury will be discountable. While sea turtles are regularly taken by hopper dredges, apparently failing to react in time to avoid the overtaking draghead, possibly because they have limited hearing abilities at lower frequencies, no sawfish take by a dredge [of any type] has ever been reported to NMFS. The implementation of NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions* may provide an additional measure of protection.

Sawfish may also be affected by blasting. Underwater explosions produce a pressure waveform with rapid oscillations from positive pressure to negative pressure that results in rapid volume changes in gas-containing organs. In fish, the swimbladder, a gas-containing organ, is the most frequently damaged organ (Christian 1973; Faulk and Lawrence 1973; Kearns and Boyd 1965; Linton et al. 1985a; Yelverton et al. 1975). It is subject to rapid contraction and overextension in response to the explosive shock waveform (Wiley et al. 1981). Species lacking swimbladders (like smalltooth sawfish) or with small swimbladders are highly resistant to explosive pressures (Aplin 1947; Fitch and Young 1948; Goertner 1994). For example, Wiley et al. (1981) and Goertner et al. (1994) noted that hogchokers (*Trinectes maculatus*), which lack swimbladders, were extremely tolerant of underwater explosions, and greatly exceeded the tolerance of any species with swimbladders that they had tested. The USACE will require the contractor to adhere to the above safety conditions related to blasting. Based on these measures and the sawfish's likely absence from the main channel areas, NMFS believes that the effects on sawfish from blasting will be discountable.

As previously noted, no sawfish have been reported and they are unlikely to be found within the existing channel area, but may be found in the mangrove areas located at the western edge of John U. Lloyd State Park. The dredging will remove approximately 1.2 acres of fringe mangroves along the east side of the existing channel. Even so, dredging near the mangrove area will only be for a portion of the overall project time. Dredging will occur a section at a time; time spent on each section depends on the amount of pre-treatment necessary and the amount of rock to be removed from each area. There is additional mangrove habitat (that will not be impacted) available directly adjacent to the project site and within the park. Smalltooth sawfish may be affected by being temporarily unable to use discreet sections affected by construction due to potential avoidance of construction activities, blasting, and related noise; however, disturbance from dredging activities and related noise in areas most likely to be used by sawfish will be intermittent, localized, and only for part of the construction period.

Like many elasmobranchs, juvenile smalltooth sawfish exhibit site fidelity to the areas in which they are pupped for the first several years of their lives, typically remaining in very shallow nearshore waters where they can avoid predation by coastal shark species. In South Florida, sawfish have established distinct nursery areas where they utilize shallow, euryhaline habitat and red mangroves for foraging and refuge; these areas have been designated as critical habitat for the species (discussed below), though NMFS expects that areas outside of the designated critical habitat are used by some sawfish for pupping and nursery habitat, where there is appropriate juvenile habitat. As noted, dredging will remove approximately 1.2 acres of fringe mangroves along the east side of the existing channel. Additional mangrove habitat exists directly adjacent to the impact area, along the west side of John U. Lloyd State Park (see Figure 4), which will be preserved. USACE will also install breaks in the riprap area to allow better access to these mangroves. NMFS believes that if juvenile sawfish are using the mangroves they are likely to be using the more suitable habitat inside the park; the 1.2 acres of fringe mangroves are in deeper waters directly adjacent to the shipping channel and as such are subject to constant traffic and pollutants. The continual vessel traffic and deeper water means these fringe mangroves are not the preferred habitat for juvenile sawfish, which prefer shallow (0-3 ft depth), undisturbed habitats. Therefore, we believe that habitat related effects on sawfish will be insignificant.

Figure 4. Mangrove impacts along the Southern Access Channel

Sea Turtles

Hydraulic and Mechanical Dredges

Large, hydraulic suction cutterhead dredges will be used to complete the deepening and widening of the channel. Smaller, mechanical clamshell-type ("bucket") dredges may also be used on portions of the project. NMFS believes the chance of injury or death from interactions with clamshell and/or hydraulic dredging equipment is discountable as these dredge types advance very slowly and sea turtles are highly mobile and are likely to avoid the areas during construction. NMFS has received very few reports of sea turtle takes associated with these dredging methods in the South Atlantic region: only 1 (live) sea turtle has been taken by a clamshell dredge over the past 33 years. The take occurred at Cape Canaveral, Florida, which routinely has very high local sea turtle abundance. Cold-stunned turtles have also been taken by cutterhead dredging, but this also rarely happens and has been generally limited to shallow, confined waters (e.g., Laguna Madre, Texas) or bays where turtles get trapped and stunned when the rapid passage of a cold front causes the temperature of the shallow water body to drop abruptly. Due to the infrequency of interactions with these gear types and the project location and channel depths, NMFS believes that the likelihood of cold stunning occurring is discountable and also that the possibility of a sea turtle being taken by a hydraulic cutterhead or a clamshell dredge is discountable.

Disposal Vessels

NMFS believes that the possibility that disposal vessel(s) will collide with and injure or kill sea turtles during disposal operations is discountable, given the vessels' slow speed (the fastest disposal scows travel at speeds of 12 knots or less (pers. comm. Terri Jordan-Sellers, USACE to Kelly Logan, NMFS, February 19, 2014), the ability of these species to move out of the way, and anticipated avoidance behavior by sea turtles at the sea surface or in the water column. Furthermore, NMFS believes the proposed dredged material (approximately 4.57 mcy) disposal activities over the life of the project are not likely to adversely affect sea turtles. Sea turtles may be attracted to ODMDS sites, to forage on the bycatch that may be occasionally found in the dredged material being dumped. As such, turtles could be potentially impacted by the sediments being discharged overhead. However, NMFS has never received a report of an injury to a sea turtle resulting from burial in, or impacts from, dredge disposal sediments, from inshore or offshore disposal sites, anywhere the USACE conducts dredged material disposal operations. Sea turtles are highly mobile and apparently are able to avoid descending dredged material discharged at the surface. NMFS believes the possibility of injury, or burial of normal, healthy sea turtles by dredged material disposal, is discountable.

Habitat Loss

Habitat effects to sea turtles include the loss of less than 8 acres of seagrass (4.67 of which is Johnson's seagrass) and some coral reef habitat. There is no nearshore hardbottom within the action area to attract foraging juvenile green turtles. However, there are some deeper reef areas within the expansion of the outer entrance channel that may contain sponges and crabs (foraged by hawksbill and loggerhead turtles, respectively). NMFS believes that foraging habitat for sea turtles is not likely a limiting factor in the action area, and thus the loss of potential seagrass and coral reef foraging habitat within the action area will have insignificant effects on sea turtles.

Blasting

Underwater explosions may affect marine life by causing death, injury, temporary threshold shifts (TTS or recoverable hearing loss), or behavioral reactions, depending on the distance an animal is located from a blast. An underwater explosion is composed of an initial shock wave, followed by a succession of oscillating bubble pulses. A shock wave is a compression wave that expands radially out from the detonation point of an explosion. At a distance from a detonation, the propagation of the shock wave may be affected by several components including the direct shock wave, the surface-reflected wave, the bottom-reflected wave, and the bottom-transmitted wave. The direct shock wave results in the peak shock pressure (compression) and the reflected wave at the air-water surface produces negative pressure (expansion). For an explosion with the same energy and at the same distance, an underwater blast is much more dangerous to animals than an air blast. The shock wave in air dissipates more rapidly and tends to be reflected at the body surface; in water the blast wave travels through the body and may cause internal injury to gas-filled organs due to impedance differences at the gas-liquid interface.

Explosions are known to injure and kill sea turtles (Duronslet et al. 1986, Gitschlag 1990, Gitschlag and Herczeg 1994, Klima et al. 1988, O'Keefe and Young 1984). NMFS studied the effects of offshore oil and gas structure removals using 23 kg (50 lb) of nitromethane (Klima et al. 1988). Caged loggerhead and Kemp's ridley sea turtles were placed at distances of 700 ft (213.4 m), 1,200 ft (365.8 m), 1,800 ft (548.6 m), and 3,000 ft (914.4 m) from the platform to be removed with explosives. The charges were placed inside platform pilings at a depth of 5 m below the mudline. Post-detonation, 4 sea turtles within1,000 ft of the explosion were unconscious, as well as an individual at 3,000 ft. Sea turtles were expected to have drowned if not recovered from the water following the detonation. All turtles exposed to the blast exhibited everted cloacas and vasodilation lasting 2-3 weeks.

The sea turtle ear appears to be adapted to both aerial and aquatic environments. Sea turtles have a primitive reptilian ear and are considered to be hearing generalists, having limited hearing abilities at lower frequencies. Although there is some variation in sea turtle hearing measurements between species and size classes (Ketten and Bartol 2006), the available data suggest that species of sea turtles are likely sensitive to frequencies from approximately 100 Hertz (Hz) to 2,000 Hz (Lenhardt 1994, Lenhardt et al. 1996, McCauley et al. 2000a and 2000b, Moein et al. 1994, O'Hara and Wilcox 1990), with greatest underwater hearing sensitivities below 1,000 Hz (Ketten and Bartol 2006). Confined underwater blasts generally produce pulses of sound at low frequencies of several Hz to a few kHz (Hall 2010). Therefore, confined blasting will likely be heard by sea turtles and may result in behavioral reactions. Behavioral reactions to the sound produced from explosions may be important if they occur in biologically important areas such as foraging areas, near nesting beaches during nesting season, or in developmental juvenile habitats. The action area is not located near any nesting beaches or known juvenile development habitats, therefore we believe that behavioral effects due to sounds produced by confined blasting will be insignificant.

For all turtle species, potential routes of effects from the use of blasting are not likely to result in adverse effects for the following reasons:

1. Blasting mitigative procedures as proposed by the USACE are detailed in Section 2. Test blasts will be performed prior to the actual project blasting. Observers will also be stationed to observe for endangered species prior to test and project blasts. Test blasts are expected to cause sea turtles to leave the project area with, at most, insignificant behavioral modifications.

2. Studies have shown that stemmed blasts have up to a 60% to 90% decrease in the strength of the pressure wave released, compared to open-water blasts of the same charge weight (Hempen et al. 2007; Hempen et al. 2005; Nedwell and Thandavamoorthy 1992). However, unlike open-water blasts, very little documentation exists on the effects that confined blasting can have on marine animals near the blast (Keevin et al. 1999). The blast mitigation procedures detailed above, in particular the rigorous observer program, have been successfully used in several recent USACE projects (i.e., San Juan Harbor, Puerto Rico, in 1994, Miami Harbor in 2005, and Wilmington Harbor in 2012).

Since these procedures have been successfully used in several recent projects without incident, it is our continued judgment that they provide sufficient protections to sea turtles, and thus the effects from blasting are discountable.

Hopper Dredge Vessel Collisions
NMFS believes that the possibility that the hopper dredge vessel(s) will collide with and injure or kill sea turtles during dredging and/or sand pumpout operations is discountable, given the vessel's slow speed, the ability of these species to move out of the way, and anticipated avoidance behavior by sea turtles at the sea surface or in the water column.

Hopper Dredge Entrainment Effects
Leatherback Sea Turtles
NMFS believes the potential use of a hopper dredge may affect, but is not likely to adversely affect, leatherback sea turtles. Leatherback sea turtles tend to be open ocean, pelagic foragers and are uncommon in shallow nearshore waters, except during nesting season or during times when they may come in towards shore to feed on aggregations of jellyfish. The project area is not located near any nesting beaches. There has never been a reported take of a leatherback by a hopper dredge. The typical leatherback would be as large as or larger than the large, industry-standard California-type hopper dredge trailing-suction draghead, making leatherbacks unlikely to be entrained. Additionally, the California-type draghead design and level position during dredging (as opposed to more upright positioning of other dredge types), makes it less likely to entrain larger sea turtles (Studt 1987). Lastly, in over 32 years of observer-monitored hopper dredging projects in Jacksonville District, only 1 leatherback was ever been reported as lethally taken or observed, and that was in a relocation trawl. Relocation trawling is not proposed for this project. Based on the above, we believe that the risk of hopper dredging effects on leatherback sea turtles will be discountable. Leatherback sea turtles will not be discussed further in this opinion.

Hawksbill Sea Turtles
Hawksbill sea turtle nesting occurs in Puerto Rico, the U.S. Virgin Islands, and along the southeast coast of Florida. Outside of the nesting areas, hawksbills have been seen off the U.S. Gulf of Mexico states and along the Eastern Seaboard as far north as Massachusetts, although sightings north of Florida are rare (NMFS and USFWS 1993). They are closely associated with coral reefs and other hardbottom habitats, but they are also found in other habitats including inlets, bays, and coastal lagoons (NMFS and USFWS 1993). The hawksbill's diet is highly specialized and consists primarily of sponges (Meylan 1999). During the past 20 years of NMFS consultations with the USACE on hopper dredging projects carried out in the Palm Beach Harbor, Port Everglades, Port of Miami, and Key West areas there has never been a documented take of a hawksbill sea turtle by a hopper dredge.[6] Due to hawksbill sea turtles' preferred habitat and diet, it is not expected that interactions would occur in the action area; therefore, NMFS believes the possibility that they would be adversely affected by hopper dredge is discountable. Hawksbill sea turtles will not be considered further in this opinion.

Kemp's Ridley Sea Turtles

[6] http://el.erdc.usace.army.mil/seaturtles/index.cfm

NMFS believes the routes of effects from the potential use of a hopper dredge may affect, but are not likely to adversely affect, Kemp's ridley sea turtles because they have not been encountered during the past 20 years of hopper dredging activities in Palm Beach Harbor, Port Everglades, Port of Miami, or Key West. This species has a very restricted range relative to other sea turtle species with most adults occurring in the Gulf of Mexico in shallow near shore waters, although adult-sized individuals sometimes are found on the eastern seaboard of the United States as well. Nesting is essentially limited to the beaches of the western Gulf of Mexico, primarily in the Mexican state of Tamaulipas, although few nests have also been recorded in Florida and the Carolinas (Meylan et al. 1995). Atlantic juveniles/subadults travel northward with vernal warming to feed in the productive, coastal waters of Georgia through New England, returning southward with the onset of winter to escape the cold (Henwood and Ogren 1987, Lutcavage and Musick 1985, Ogren 1989). Upon leaving Chesapeake Bay in autumn, juvenile ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined there by juveniles of the same size from North Carolina sounds and smaller juveniles from New York and New England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly et al. 1995c, Epperly et al. 1995b, Musick and Limpus 1997). Adult Kemp's ridleys primarily occupy neritic habitats, typically containing muddy or sandy bottoms where prey can be found. In the post-pelagic stages, Kemp's ridley sea turtles are largely cancrivorous (crab eating), with a preference for portunid (swimming) crabs (Bjorndal 1997). Stomach contents of Kemp's ridleys along the lower Texas coast consisted of a predominance of nearshore crabs and mollusks, as well as fish, shrimp, and other foods considered to be scavenged discards from the shrimping industry (Shaver 1991). Kemp's ridley sea turtles will not be considered further in this opinion based on the improbability of their presence in the action area and a low likelihood of an encounter with a hopper dredge.

Humpback Whales

Humpback whales may be found in or near the action area. These species are generally found seaward of the continental shelf, and would only be in the action area during migrations to and from breeding grounds (during the spring and fall months). NMFS has analyzed the routes of potential effects on humpback whales from the proposed action and, based on our analysis, determined that potential effects are limited to the following: injury from potential interactions with construction equipment (e.g., a dredge vessel striking a whale), injury from use of explosives, and temporary avoidance of the area during construction operations. The USACE will require the contractor to follow the aforementioned blasting safety conditions. Blasting would result in temporary impacts and would not be a daily occurrence of the project. In addition, whales do not use this area throughout the year and would most likely be migrating, the USACE would not be blasting during a large portion of the year (November through March), as per the requirements to avoid harm to manatees listed above. Therefore, NMFS concludes that the project's blasting effects are discountable.

In addition, the dredge crew and contractors will be required to abide by NMFS's *Vessel Strike Avoidance and Reporting Guidelines* (Appendix A) and all dredges will be required to have NMFS-approved endangered species observers aboard. NMFS believes that the possibility that the disposal vessel(s) will collide with and injure or kill whales during disposal operations is discountable, given the vessel's slow speed (the fastest disposal scows travel at speeds of 12 knots or less (pers. comm. Terri Jordan-Sellers, USACE to Kelly Logan, NMFS, February 19, 2014), and the whales' limited, seasonal presence in the action area. With implementation of these conservation measures, NMFS believes that the likelihood of a dredge or disposal vessel striking a humpback whale is discountable.

Sperm Whales

Sperm whales are predominantly found seaward of the continental shelf and are not expected to be found within the shallow waters inshore of the outer reef nor at the ODMDS. Therefore, we believe the risk to sperm whales from blasting or dredging impacts, including potential collision with a dredge vessel en route to or from the ODMDS, is discountable.

Proposed Critical Habitat for Loggerhead Sea Turtles

While a portion of the project occurs in proposed critical habitat for loggerheads, specifically unit Logg-N-19, which includes concentrated breeding habitat and constricted migratory corridor habitat, the project is not expected to impact the primary constituent elements (PCEs) and thus the habitat itself. The PCEs that support breeding habitat are (1) high concentrations of reproductive male and female loggerheads; (2) proximity to primary Florida migratory corridor; and (3) proximity to Florida nesting grounds. The PCEs for constricted migratory habitat are (1) constricted continental shelf area relative to nearby continental shelf waters that concentrate migratory pathways; and (2) passage conditions to allow for migration to and from nesting, breeding, and/or foraging areas. Dredging and port expansion will not alter the PCEs for breeding habitat as it will not impact the high concentration of reproductive individuals in the area nor the proximity to the nesting grounds or migratory corridor. The PCEs for the constricted migratory corridor will not be impacted as the project will not alter the passage conditions of the corridor. Therefore, effects to loggerhead critical habitat as it is currently proposed are discountable.

4.2 Species and Critical Habitat Likely to be Adversely Affected

NMFS believes that the proposed project may affect green and loggerhead sea turtles, Johnson's seagrass, staghorn coral, 6 coral species proposed to be listed, and elkhorn and staghorn coral-designated critical habitat.

4.2.1 Sea Turtles

The following subsections are synopses of the best available information on the status of the sea turtle species that are likely to be adversely affected by 1 or more components of the proposed action, including information on the distribution, population structure, life history, abundance, and population trends of each species and threats to each species. The biology and ecology of these species as well as their status and trends inform the effects analysis for this opinion. Additional background information on the status of sea turtle species can be found in a number of published documents, including: recovery plans for the Atlantic green sea turtle (NMFS and USFWS 1991), and loggerhead sea turtle (NMFS and USFWS 2008a); Pacific sea turtle recovery plans (NMFS and USFWS 1998a; NMFS and USFWS 1998b; NMFS and USFWS 1998c; NMFS and USFWS 1998b); and sea turtle status reviews, stock assessments, and biological reports (Conant et al. 2009; NMFS-SEFSC 2001; NMFS-SEFSC 2009a; NMFS and USFWS 1995b; NMFS and USFWS 2007a; NMFS and USFWS 2007b; NMFS and USFWS 2007c;

NMFS and USFWS 2007d; NMFS and USFWS 2007e; TEWG 1998; TEWG 2000a; TEWG 2007; TEWG 2009).

4.2.1.1 General Threats Faced by All Sea Turtle Species

Sea turtles face numerous natural and anthropogenic threats that shape their status and affect their ability to recover. As many of the threats are the same or similar in nature for all listed sea turtle species, those identified in this section are discussed in a general sense for all listed sea turtles. Threat information specific to a particular species is then discussed in the corresponding status section where appropriate.

Fisheries

Incidental bycatch in commercial fisheries is identified as a major contributor to past declines, and threat to future recovery, for all of the sea turtle species (NMFS and USFWS 1991, 1992, 1993, 2008, 2011). Domestic fisheries often capture, injure, and kill sea turtles at various life stages. Sea turtles in the pelagic environment are exposed to U.S. Atlantic pelagic longline fisheries. Sea turtles in the benthic environment in waters off the coastal United States are exposed to a suite of other fisheries in federal and state waters. These fishing methods include trawls, gillnets, purse seines, hook-and-line gear [including bottom longlines and vertical lines (e.g., bandit gear, handlines, and rod-reel)], pound nets, and trap fisheries. (Refer to the Environmental Baseline section of this opinion for more specific information regarding federal and state managed fisheries affecting sea turtles within the action area). The Southeast shrimp fisheries have historically been the largest fishery threat to benthic sea turtles in the southeastern United States, and continue to interact with and kill large numbers of sea turtles each year.

In addition to domestic fisheries, sea turtles are subject to direct as well as incidental capture in numerous foreign fisheries, further impeding the ability of sea turtles to survive and recover on a global scale. For example, pelagic stage sea turtles, especially loggerheads and leatherbacks, circumnavigating the Atlantic are susceptible to international longline fisheries including the Azorean, Spanish, and various other fleets (Aguilar et al. 1995; Bolten et al. 1994; Crouse 1999). Bottom longline and gillnet fishing are known to occur in many foreign waters, including (but not limited to) the northwest Atlantic, western Mediterranean, South America, West Africa, Central America, and the Caribbean. Shrimp trawl fisheries are also operating off the shores of numerous foreign countries and pose a significant threat to sea turtles similar to the impacts seen in U.S. waters. Many unreported takes or incomplete records by foreign fleets make it difficult to characterize the total impact that international fishing pressure is having on listed sea turtles. Nevertheless, international fisheries represent a continuing threat to sea turtle survival and recovery throughout their respective ranges.

Non-Fishery In-Water Activities

There are also many non-fishery impacts affecting the status of sea turtle species, both in the ocean and on land. In nearshore waters of the United States, the construction and maintenance of federal navigation channels has been identified as a source of sea turtle mortality. Hopper dredges, which are frequently used in ocean bar channels and sometimes in harbor channels and offshore borrow areas, move relatively rapidly and can entrain and kill sea turtles (NMFS 1997a). Sea turtles entering coastal or inshore areas have also been affected by entrainment in the cooling-water systems of electrical generating plants. Other nearshore threats include harassment and/or injury resulting from private and commercial vessel operations, military detonations and training exercises, in-water construction activities, and scientific research activities.

Coastal Development and Erosion Control

Coastal development can deter or interfere with nesting, affect nesting success, and degrade nesting habitats for sea turtles. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may decrease the amount of nesting area available to females and change the natural behaviors of both adults and hatchlings, directly or indirectly, through loss of beach habitat or changing thermal profiles and increasing erosion, respectively. (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). In addition, coastal development is usually accompanied by artificial lighting which can alter the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings that are drawn away from the water (Witherington and Bjorndal 1991). In-water erosion control structures such as breakwaters, groins, and jetties can impact nesting females and hatchlings as they approach and leave the surf zone or head out to sea by creating physical blockage, concentrating predators, creating longshore currents, and disrupting of wave patterns.

Environmental Contamination

Multiple municipal, industrial, and household sources, as well as atmospheric transport, introduce various pollutants such as pesticides, hydrocarbons, organochlorides (e.g., DDT, PCBs, and PFCs), and others that may cause adverse health effects to sea turtles (Garrett 2004; Grant and Ross 2002; Hartwell 2004; Iwata et al. 1993). Acute exposure to hydrocarbons from petroleum products released into the environment via oil spills and other discharges may directly injure individuals through skin contact with oils (Geraci 1990), inhalation at the water's surface and ingesting compounds while feeding (Matkin and Saulitis 1997). Hydrocarbons also have the potential to impact prey populations, and therefore may affect listed species indirectly by reducing food availability in the action area. In 2010, there was a massive oil spill in the Gulf of Mexico at BP's Macondo well. Official estimates are that millions of barrels of oil were released into the Gulf of Mexico. Additionally, approximately 1.8 million gallons of chemical dispersant were applied on the seawater surface and at the wellhead to attempt to break down the oil. At this time the assessment of total direct impact to sea turtles has not been determined. Additionally, we do not know the long-term impacts to sea turtles because of habitat impacts, prey loss, and subsurface oil particles and oil components broken down through physical, chemical, and biological processes.

Marine debris is a continuing problem for sea turtles. Sea turtles living in the pelagic environment commonly eat or become entangled in marine debris (e.g., tar balls, plastic bags/pellets, balloons, and ghost fishing gear) as they feed along oceanographic fronts where debris and their natural food items converge. This is especially problematic for sea turtles that spend all or significant portions of their life cycle in the pelagic environment (i.e., leatherbacks, juvenile loggerheads, and juvenile green turtles).

Climate Change

There is a large and growing body of literature on past, present, and future impacts of global climate change, exacerbated and accelerated by human activities. Some of the likely effects commonly mentioned are sea level rise, increased frequency of severe weather events, and change in air and water temperatures. NOAA's climate information portal provides basic background information on these and other measured or anticipated effects (see http://www.climate.gov).

Climate change impacts on sea turtles currently cannot be predicted with any degree of certainty; however, significant impacts to the hatchling sex ratios of sea turtles may result (NMFS and USFWS 2007c). In sea turtles, sex is determined by the ambient sand temperature (during the middle third of incubation) with female offspring produced at higher temperatures and males at lower temperatures within a thermal tolerance range of 25°-35°C (Ackerman 1997). Increases in global temperature could potentially skew future sex ratios toward higher numbers of females (NMFS and USFWS 2007c).

The effects from increased temperatures may be intensified on developed nesting beaches where shoreline armoring and construction have denuded vegetation. Erosion control structures could potentially result in the permanent loss of nesting beach habitat or deter nesting females (NRC 1990). These impacts will be exacerbated by sea level rise. If females nest on the seaward side of the erosion control structures, nests may be exposed to repeated tidal overwash (NMFS and USFWS 2007c). Sea level rise from global climate change is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor, as the sea may inundate nesting sites and decrease available nesting habitat (Baker et al. 2006; Daniels et al. 1993; Fish et al. 2005). The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Antonelis et al. 2006; Baker et al. 2006).

Other changes in the marine ecosystem caused by global climate change (e.g., ocean acidification, salinity, oceanic currents, dissolved oxygen levels, nutrient distribution, etc.) could influence the distribution and abundance of lower trophic levels (e.g., phytoplankton, zooplankton, submerged aquatic vegetation, crustaceans, mollusks, forage fish, etc.) which could ultimately affect the primary foraging areas of sea turtles.

Other Threats

Predation by various land predators is a threat to developing nests and emerging hatchlings. The major natural predators of sea turtle nests are mammals, including raccoons, dogs, pigs, skunks, and badgers. Emergent hatchlings in the United States are preyed upon by these mammals as well as ghost crabs, laughing gulls, and the exotic South American fire ant (*Solenopsis invicta*). In addition to natural predation, direct harvest of eggs and adults from beaches in foreign countries continues to be a problem for various sea turtle species throughout their ranges (NMFS and USFWS 2008a).

Diseases, toxic blooms from algae and other microorganisms, and cold stunning events are additional sources of mortality that can range from local and limited to wide-scale and impacting hundreds or thousands of animals.

Actions Taken to Reduce Threats

Actions have been taken to reduce anthropogenic impacts to sea turtles from various sources, particularly since the early 1990s. These include lighting ordinances, predation control, and nest relocations to help increase hatchling survival, as well as measures to reduce the mortality of pelagic immatures, benthic immatures, and sexually mature age classes from various fisheries and other marine activities. Some actions have resulted in significant steps towards reducing the recurring sources of mortality of sea turtles in the environmental baseline and improving the status of all sea turtle populations in the Atlantic and Gulf of Mexico. For example, the TED regulation published on February 21, 2003 (68 FR 8456), represent a significant improvement in the baseline effects of trawl fisheries on sea turtles, though shrimp trawling is still considered to be one of the largest source of anthropogenic mortality for most of our sea turtle species (NMFS-SEFSC 2009a).

4.2.1.2 Loggerhead Sea Turtle – Northwest Atlantic DPS

The loggerhead sea turtle was listed as a threatened species throughout its global range on July 28, 1978. NMFS and USFWS subsequently published a final rule listing 9 DPSs of loggerhead sea turtles (76 FR 58868, September 22, 2011, effective October 24, 2011). The DPSs established by this rule include (1) Northwest Atlantic Ocean (threatened), (2) Northeast Atlantic Ocean (endangered), (3) South Atlantic Ocean (threatened), (4) Mediterranean Sea (endangered), (5) North Pacific Ocean (endangered), (6) South Pacific Ocean (endangered), (7) North Indian Ocean (endangered), (8) Southeast Indo-Pacific Ocean (endangered), and (9) Southwest Indian Ocean (threatened). The Northwest Atlantic Ocean (NWA) DPS is the only 1 that occurs within the action area and therefore is the only one considered in this Opinion. NMFS has proposed to designate critical habitat for the NWA DPS of loggerhead sea turtles. Specific areas proposed for designation include 36 occupied marine areas within the range of the NWA DPS. These areas contain one or a combination of nearshore reproductive habitat, winter area, breeding areas, and migratory corridors. The rule is scheduled to be finalized in July 2014.

Species Description and Distribution

Loggerheads are large sea turtles with the mean straight carapace length (SCL) of adults in the southeast United States being approximately 3 ft (92 cm). The corresponding mass is approximately 255 lb (116 kg) (Ehrhart and Yoder 1978). Adult and subadult loggerhead sea turtles typically have a light yellow plastron and a reddish brown carapace covered by non-overlapping scutes that meet along seam lines. They typically have 11 or 12 pairs of marginal scutes, 5 pairs of costals, 5 vertebrals, and a nuchal (precentral) scute that is in contact with the first pair of costal scutes (Dodd 1988).

The loggerhead sea turtle inhabits continental shelf and estuarine environments throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd 1988). Habitat uses within these areas vary by life stage. Juveniles are omnivorous and forage on crabs, mollusks, jellyfish and vegetation at or near the surface (Dodd 1988). Subadult and adult loggerheads are primarily found in coastal waters and prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom habitats.

The majority of loggerhead nesting occurs at the western rims of the Atlantic and Indian Oceans concentrated in the north and south temperate zones and subtropics (NRC 1990). In the western North Atlantic, loggerhead nesting is concentrated along the coasts of the United States from southern Virginia to Alabama. Additional nesting beaches are found along the northern and western Gulf of Mexico, eastern Yucatán Peninsula, at Cay Sal Bank in the eastern Bahamas (Addison 1997; Addison and Morford 1996), off the southwestern coast of Cuba (Gavilan 2001), and along the coasts of Central America, Colombia, Venezuela, and the eastern Caribbean Islands.

Non-nesting, adult female loggerheads are reported throughout the United States and Caribbean Sea. Little is known about the distribution of adult males, which are seasonally abundant near nesting beaches. However, aerial surveys suggest that the general species distribution of loggerheads in U.S. waters is as follows: 54% in the Atlantic off the southeast United States, Atlantic, 29% in the Atlantic off the northeast United States, 12% in the eastern Gulf of Mexico, and 5% in the western Gulf of Mexico (TEWG 1998).

Within the NWA, most loggerhead sea turtles nest from North Carolina to Florida and along the Gulf coast of Florida. NMFS has previously recognized at least 5 Western Atlantic subpopulations based on nesting beach assemblages, divided geographically as follows:

> (1) a Northern nesting subpopulation, occurring from North Carolina to Northeast Florida at about 29ºN;
>
> (2) a South Florida nesting subpopulation, occurring from 29°N on the east coast of the state to Sarasota on the west coast;
>
> (3) a Florida Panhandle nesting subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida;
>
> (4) a Yucatán nesting subpopulation, occurring on the Eastern Yucatán Peninsula, Mexico (Márquez M 1990; TEWG 2000a); and
>
> (5) a Dry Tortugas nesting subpopulation, occurring in the islands of the Dry Tortugas, near Key West, Florida (NMFS-SEFSC 2001).

The recovery plan for the Northwest Atlantic population of loggerhead sea turtles concluded, based on recent advances in genetic analyses, that there is no genetic distinction between loggerheads nesting on adjacent beaches along the Florida Peninsula and that specific boundaries for subpopulations could not be designated based on genetic differences alone. Thus, the plan uses a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to identify recovery units. The recovery units are (1) the Northern Recovery Unit (Florida/Georgia border north through southern Virginia), (2) the Peninsular Florida Recovery Unit (Florida/Georgia border through

Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (Franklin County, Florida, through Texas), and (5) the Greater Caribbean Recovery Unit (Mexico through French Guiana, the Bahamas, Lesser Antilles, and Greater Antilles) (NMFS and USFWS 2008a). The recovery plan concluded that all recovery units are essential to the recovery of the species. Although the recovery plan was written prior to the listing of the NWA DPS, the recovery units for what was then termed the Northwest Atlantic population apply to the NWA DPS.

Life History Information

The Northwest Atlantic Loggerhead Recovery Team defined the following 8 life stages for the loggerhead life cycle, including the ecosystems those stages generally use: (1) egg (terrestrial zone), (2) hatchling stage (terrestrial zone), (3) hatchling swim frenzy and transitional stage (neritic zone[7]), (4) juvenile stage (oceanic zone), (5) juvenile stage (neritic zone), (6) adult stage (oceanic zone), (7) adult stage (neritic zone), and (8) nesting female (terrestrial zone) (NMFS and USFWS 2008). Loggerheads are long-lived organisms that reach sexual maturity between 20 and 38 years of age, although this varies widely among populations (Frazer and Ehrhart 1985; NMFS and SEFSC 2001). The annual mating season for loggerhead sea turtles occurs from late March to early June, and eggs are laid throughout the summer months. Female loggerheads deposit an average of 4.1 nests within a nesting season (Murphy and Hopkins 1984) but an individual female only nests every 3.7 years on average (Tucker 2010). Along the southeastern United States, loggerheads lay an average of 100 to 126 eggs per nest (Dodd 1988) which incubate for 42 to 75 days before hatching (NMFS and USFWS 2008b).

As post-hatchlings, loggerheads hatched on U.S. beaches migrate offshore and become associated with *Sargassum* habitats, driftlines, and other convergence zones (Carr 1986), (Witherington 2002). Loggerheads originating from the NWA DPS are believed to lead a pelagic existence in the North Atlantic Gyre for a period as long as 7-12 years (Bolten et al. 1998) before moving to more coastal habitats. Recent studies have suggested that not all loggerhead sea turtles follow the model of circumnavigating the North Atlantic Gyre as pelagic juveniles, followed by permanent settlement into benthic environments (Bolten and Witherington 2003; Laurent et al. 1998). These studies suggest some turtles may either remain in the pelagic habitat in the North Atlantic longer than hypothesized or move back and forth between pelagic and coastal habitats interchangeably (Witzell 2002). Stranding records indicate that when immature loggerheads reach 15-24 inches (40-60 cm) SCL, they begin to occur in coastal inshore waters of the continental shelf throughout the U.S. Atlantic and Gulf of Mexico (Witzell 2002).

After departing the oceanic zone, neritic juvenile loggerheads in the Northwest Atlantic inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Estuarine waters of the United States, including areas such as Long Island Sound, Chesapeake Bay, Pamlico and Core Sounds, Mosquito and Indian River Lagoons, Biscayne Bay, Florida Bay, and numerous embayments fringing the Gulf of Mexico, comprise important inshore habitat. Along the Atlantic and Gulf of Mexico shoreline, essentially all shelf waters are inhabited by loggerheads.

---

[7] Neritic refers to the inshore marine environment from the surface to the sea floor where water depths do not exceed 200 meters.

Like juveniles, non-nesting adult loggerheads also use the neritic zone. However, these adult loggerheads use the relatively enclosed shallow-water estuarine habitats with limited ocean access are less frequently than the juveniles. Juveniles, but not adult loggerheads, regularly use areas such as Pamlico Sound, North Carolina, and the Indian River Lagoon, Florida. In comparison, adult loggerheads tend to use estuarine areas with more open ocean access, such as Chesapeake Bay in the U.S. Mid-Atlantic. Shallow-water habitats with large expanses of open ocean access, such as Florida Bay, provide year-round resident foraging areas for significant numbers of male and female adult loggerheads. Offshore, adults primarily inhabit continental shelf waters, from New York south through Florida, The Bahamas, Cuba, and the Gulf of Mexico. Seasonal use of Mid-Atlantic shelf waters, especially offshore New Jersey, Delaware, and Virginia during summer months, and offshore shelf waters, such as Onslow Bay (off the North Carolina coast), during winter months has also been documented (Hawkes et al. 2007a; Georgia Department of Natural Resources, unpublished data; South Carolina Department of Natural Resources, unpublished data). Satellite telemetry has identified the shelf waters along the west Florida coast, The Bahamas, Cuba, and the Yucatán Peninsula as important resident areas for adult female loggerheads that nest in Florida (Foley et al. 2008; M. Lamont, Florida Cooperative Fish and Wildlife Research Unit, personal communication, 2009; M. Nicholas, National Park Service, personal communication, 2009). The southern edge of the Grand Bahama Bank is important habitat for loggerheads nesting on the Cay Sal Bank in The Bahamas, but nesting females are also resident in the bights of Eleuthera, Long Island, and Ragged Islands as well as Florida Bay in the United States, and the north coast of Cuba (A. Bolten and K. Bjorndal, University of Florida, unpublished data). Moncada et al. (2009) report the recapture in Cuban waters of 5 adult female loggerheads originally flipper tagged in Quintana Roo, Mexico, indicating that Cuban shelf waters likely also provide foraging habitat for adult females that nest in Mexico.

Status and Population Dynamics

A number of stock assessments and similar reviews (Conant et al. 2009; Heppell et al. 2003a; NMFS-SEFSC 2009a; NMFS and SEFSC 2001; NMFS and USFWS 2008a; TEWG 1998; TEWG 2000a; TEWG 2009) have examined the stock status of loggerheads in the Atlantic Ocean, but none have been able to develop a reliable estimate of absolute population size.

Numbers of nests and nesting females can vary widely from year to year. However, nesting beach surveys can provide a reliable assessment of trends in the adult female population, due to the strong nest site fidelity of female loggerhead sea turtles, as long as such studies are sufficiently long and effort and methods are standardized [see, e.g., NMFS and USFWS (2008a)]. NMFS and USFWS (2008a) concluded that the lack of change in two important demographic parameters of loggerheads, remigration interval and clutch frequency, indicate that time series on numbers of nests can provide reliable information on trends in the female population.

<u>Peninsular Florida Recovery Unit</u>

The Peninsular Florida Recovery Unit (PFRU) is the largest loggerhead nesting assemblage in the Northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females per year (NMFS and USFWS 2008a). The statewide estimated total for 2012 was 98,601 nests (FWRI nesting database).

In addition to the total nest count estimates, the Florida Fish and Wildlife Research Institute (FWRI) uses an index nesting beach survey method. The index survey uses standardized data-collection criteria to measure seasonal nesting and allow accurate comparisons between beaches and between years. This provides a better tool for understanding the nesting trends (Figure 5). FWRI performed a detailed analysis of the long-term loggerhead index nesting data (1989-2012) (http://myfwc.com/research/wildlife/sea-turtles/nesting/loggerhead-trends/). Three distinct trends over that time period were identified. From 1989-1998 there was a 23% increase, that was then followed by a sharp decline over the subsequent decade. However, recent large increases in loggerhead nesting occurred since then. FWRI examined the trend from the 1998 nesting high through 2012 and found the decade-long post-1998 decline had reversed and there was no longer a demonstrable trend. Looking at the data from 1989 through 2012 FWRI concluded that there was an overall positive change in the nest counts.

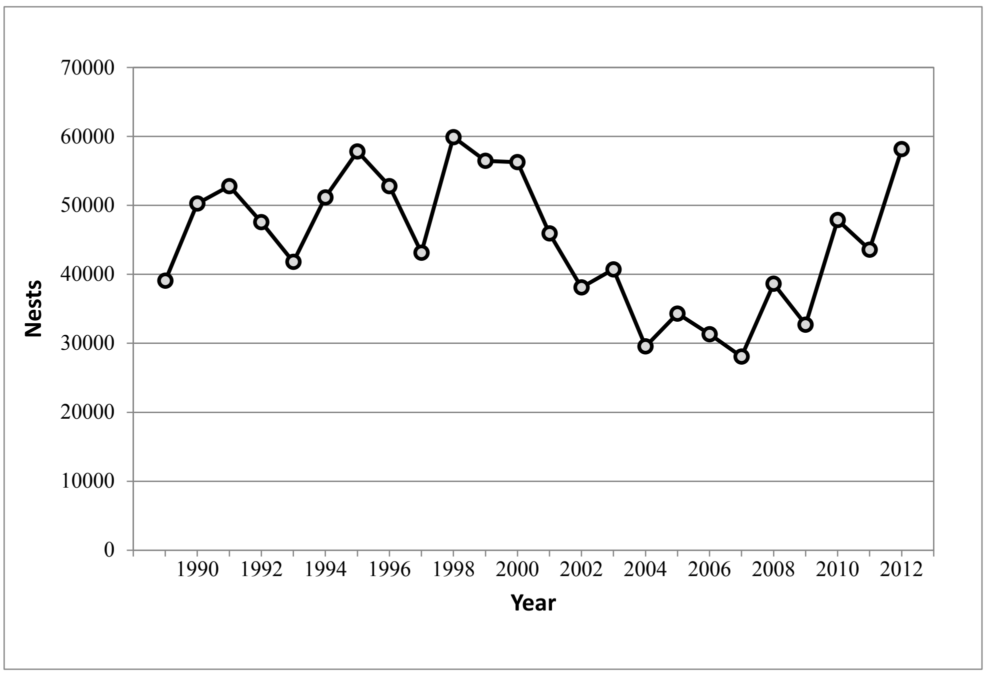

Figure 5. Loggerhead sea turtle nesting at Florida index beaches since 1989

Northern Recovery Unit

Annual nest totals from beaches within the Northern Recovery Unit (NRU) averaged 5,215 nests from 1989-2008, a period of near-complete surveys of NRU nesting beaches (Georgia Department of Natural Resources (GDNR) unpublished data, North Carolina Wildlife Resources Commission (NCWRC) unpublished data, South Carolina Department of Natural Resources (SCDNR) unpublished data), and represent approximately 1,272 nesting females per year, assuming 4.1 nests per female (Murphy and Hopkins 1984). The loggerhead nesting trend from daily beach surveys showed a significant decline of 1.3% annually from 1989-2008. Nest totals from aerial surveys conducted by SCDNR showed a 1.9% annual decline in nesting in South

Carolina from 1980 through 2008. Overall, there is strong statistical data to suggest the NRU had experienced a long-term decline over that period.

Data since that analysis (Table 2) are showing improved nesting numbers and a departure from the declining trend. Georgia nesting has rebounded to show the first statistically significant increasing trend since comprehensive nesting surveys began in 1989 (Mark Dodd, GADNR press release, http://www.georgiawildlife.com/node/3139). South Carolina and North Carolina nesting have also begun to show a shift away from the past declining trend.

Table 2. Total Number of NRU Loggerhead Nests (GADNR, SCDNR, and NCWRC Nesting Datasets)

Nests Recorded	2008	2009	2010	2011	2012
Georgia	1,649	997	1,761	1,992	2,218
South Carolina	4,500	2,183	3,141	4,015	4,615
North Carolina	841	276	846	948	1,069
Total	6,990	3,456	5,748	6,955	7,902

South Carolina also conducts an index beach nesting survey similar to the one described for Florida. Although the survey only includes a subset of nesting, the standardized effort and locations allow for a better representation of the nesting trend over time. Increases in nesting were seen for the period from 2009-2012, with 2012 showing the highest index nesting total since the start of the program (Figure 6).

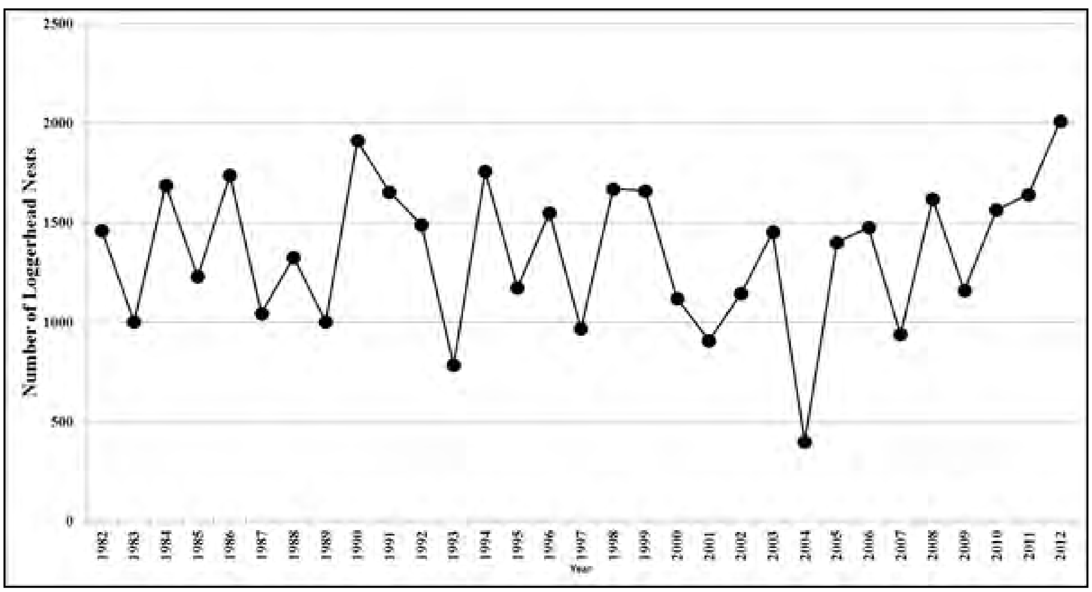

Figure 6. South Carolina index nesting beach counts for loggerhead sea turtles (from the SCDNR website, http://www.dnr.sc.gov/seaturtle/nest.htm)

<u>Other Northwest Atlantic DPS Recovery Units</u>

The remaining 3 recovery units—Dry Tortugas (DTRU), Northern Gulf of Mexico (NGMRU), and Greater Caribbean (GCRU)—are much smaller nesting assemblages but still considered essential to the continued existence of the species. Nesting surveys for the DTRU are conducted as part of Florida's statewide survey program. Survey effort was relatively stable during the 9-year period from 1995-2004 (although the 2002 year was missed). Nest counts ranged from 168-270, with a mean of 246, but with no detectable trend during this period (NMFS and USFWS 2008a). Nest counts for the NGMRU are focused on index beaches rather than all beaches where nesting occurs. Analysis of the 12-year dataset (1997-2008) of index nesting beaches in the area shows a statistically significant declining trend of 4.7% annually (NMFS and USFWS 2008a). Nesting on the Florida Panhandle index beaches, which represents the majority of NGMRU nesting, had shown a large increase in 2008, but then declined again in 2009 and 2010 before rising back to a level similar to the 2003-2007 average in 2011. Nesting survey effort has been inconsistent among the GCRU nesting beaches and no trend can be determined for this subpopulation. Zurita et al. (2003) found a statistically significant increase in the number of nests on 7 of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001, and the previously reported increasing trend appears to not have been sustained (NMFS and USFWS 2008a).

<u>In-water Trends</u>

Nesting data are the best current indicator of sea turtle population trends; however, in-water data also provide some insight. Such research suggests the abundance of neritic juvenile loggerheads is steady or increasing. Although Ehrhart et al. (2007) found no significant regression-line trend in a long-term dataset, researchers have observed notable increases in catch per unit effort (CPUE) over the past several years (Ehrhart et al. 2007, Epperly et al. 2007, Arendt et al. 2009). Researchers believe that this increase in CPUE is likely linked to an increase in juvenile abundance, though it is unclear whether this increase in abundance represents a true population increase among juveniles or merely a shift in spatial occurrence. Bjorndal et al. (2005), (cited in NMFS and USFWS (2008a), caution about extrapolating localized in-water trends to the broader population and relating localized trends in neritic sites to population trends at nesting beaches. The apparent overall increase in the abundance of neritic loggerheads in the southeastern United States may be due to increased abundance of the largest oceanic/neritic juveniles (historically referred to as small benthic juveniles), which could indicate a relatively large number of individuals around the same age may mature in the near future (TEWG 2009). However, in-water studies throughout the eastern United States also indicate a substantial decrease in the abundance of the smallest oceanic/neritic juvenile loggerheads, a pattern corroborated by stranding data (TEWG 2009).

<u>Population Estimate</u>

The NMFS Southeast Fishery Science Center developed a preliminary stage/age demographic model to help determine the estimated impacts of mortality reductions on loggerhead sea turtle population dynamics (NMFS-SEFSC 2009a). The model uses the range of published information for the various parameters including mortality by stage, stage duration (years in a stage), and fecundity parameters such as eggs per nest, nests per nesting female, hatchling emergence success, sex ratio, and remigration interval. Resulting trajectories of model runs for each individual recovery unit, as well as the western North Atlantic population as a whole, were found to be very similar. The model run estimates from the 2004-2008 time frame, suggest the adult female population size in the western North Atlantic is approximately 20,000 to 40,000 individuals, with a low likelihood of being up to 70,000 (NMFS-SEFSC 2009a). A less robust estimate for total benthic females in the western North Atlantic was also obtained, yielding approximately 30,000-300,000 individuals, up to less than 1 million (NMFS-SEFSC 2009a).

Threats

The threats faced by loggerhead sea turtles are well-summarized in the general discussion of threats in Section 4.2.1.1. However, the impact of fishery interactions is a point of further emphasis for this species. The Loggerhead Biological Review Team determined that the greatest threats to the NWA DPS of loggerheads result from cumulative fishery bycatch in neritic and oceanic habitats (Conant et al. 2009). Significant mortality occurs in longline fisheries, bottom and mid-water trawl fisheries, dredge fisheries, gillnet fisheries, and pot/trap fisheries. Although total mortality from all fisheries has not been estimated, the combined mortalities are likely significant.

Regarding the impacts of pollution, loggerheads may be particularly affected by organochlorine contaminants as they were observed in a study by Storelli et al. (2008), to have the highest organochlorine concentrations in sampled tissues (Storelli et al. 2008). Storelli et al. (2008) analyzed tissues from stranded loggerhead sea turtles and found that mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals, and porpoises (Law et al. 1991). It is thought that dietary preferences were likely to be the main differentiating factor leading to different contaminant concentrations among species.

Specific information regarding potential climate change impacts on loggerheads is also available. Future surface temperature increases of 2°–3°C are expected by 2100 (Hansen et al., 2006). Modeling suggests an increase of 2°C in air temperature would result in a sex ratio of over 80% female offspring for loggerheads nesting near Southport, North Carolina from current ratios of 50%-65%. The same increase in air temperatures at nesting beaches in Cape Canaveral, Florida, would result in close to 100% female offspring from current ratios of 90%. Such highly skewed sex ratios could undermine the reproductive capacity of the species. More ominously, an air temperature increase of 3°C is likely to exceed the thermal threshold of most clutches, leading to death (Hawkes et al. 2007). Warmer sea surface temperatures have also been correlated with an earlier onset of loggerhead nesting in the spring (Hawkes et al. 2007; Weishampel et al. 2004), as well as short inter-nesting intervals (Hays et al. 2002) and shorter nesting season (Pike et al. 2006).

4.2.1.3 Green Sea Turtle

The green sea turtle was listed as threatened under the ESA on July 28, 1978, except for the Florida and Pacific coast of Mexico breeding populations, which were listed as endangered.

Species Description and Distribution

The green sea turtle is the largest of the hardshell marine turtles, growing to a weight of 350 lb (159 kg) and a straight carapace length of greater than 3.3 ft (1 m). Green sea turtles have a smooth carapace with 4 pairs of lateral (or costal) scutes and a single pair of elongated prefrontal scales between the eyes. They typically have a black dorsal surface and a white ventral surface, although the carapace of green sea turtles in the Atlantic Ocean has been known to change in color from solid black to a variety of shades of grey, green, or brown and black in starburst or irregular patterns (Lagueux 2001).

With the exception of post-hatchlings, green sea turtles live in nearshore tropical and subtropical waters where they generally feed on marine algae and seagrasses. They have specific foraging grounds and may make large migrations between these forage sites and natal beaches for nesting (Hays et al. 2001). Green sea turtles nest on sandy beaches of mainland shores, barrier islands, coral islands, and volcanic islands in more than 80 countries worldwide (Hirth and USFWS 1997). The two largest nesting populations are found at Tortuguero, on the Caribbean coast of Costa Rica, and Raine Island, on the Pacific coast of Australia along the Great Barrier Reef. Differences in mitochondrial DNA properties of green sea turtles from different nesting regions indicate there are genetic subpopulations (Bowen et al. 1992; Fitzsimmons et al. 2006). Despite the genetic differences, sea turtles from separate nesting origins are commonly found mixed together on foraging grounds throughout the species' range. However, such mixing occurs at extremely low levels in Hawaiian foraging areas, perhaps making this central Pacific population the most isolated of all green sea turtle populations occurring worldwide (Dutton et al. 2008).

In U.S. Atlantic and Gulf of Mexico waters, green sea turtles are distributed in inshore and nearshore waters from Texas to Massachusetts. Principal benthic foraging areas in the southeastern United States include Aransas Bay, Matagorda Bay, Laguna Madre, and the Gulf inlets of Texas (Doughty 1984; Hildebrand 1982; Shaver 1994), the Gulf of Mexico off Florida from Yankeetown to Tarpon Springs (Caldwell and Carr 1957; Carr 1984), Florida Bay and the Florida Keys (Schroeder and Foley 1995), the Indian River Lagoon system in Florida (Ehrhart 1983), and the Atlantic Ocean off Florida from Brevard through Broward Counties (Guseman and Ehrhart 1992; Wershoven and Wershoven 1992). The summer developmental habitat for green sea turtles also encompasses estuarine and coastal waters from North Carolina to as far north as Long Island Sound (Musick and Limpus 1997). Additional important foraging areas in the western Atlantic include the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, scattered areas along Colombia and Brazil (Hirth 1971), and the northwestern coast of the Yucatan Peninsula.

The complete nesting range of green sea turtles within the southeastern United States includes sandy beaches between Texas and North Carolina, as well as the USVI and Puerto Rico (Dow et al. 2007; NMFS and USFWS 1991). However, the vast majority of green sea turtle nesting within the southeastern United States occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Principal U.S. nesting areas for green sea turtles are in eastern Florida, predominantly Brevard through Broward counties. For more information on green sea turtle nesting in other ocean basins, refer to the 1991 Recovery Plan for the Atlantic Green Turtle (NMFS and USFWS 1991) or the 2007 Green Sea Turtle 5-Year Status Review (NMFS and USFWS 2007a).

Life History Information

Green sea turtles reproduce sexually, and mating occurs in the waters off nesting beaches. Mature females return to their natal beaches (i.e., the same beaches where they were born) to lay eggs (Balazs 1982; Frazer and Ehrhart 1985) every 2-4 years while males are known to reproduce every year (Balazs 1983). In the southeastern United States, females generally nest between June and September, and peak nesting occurs in June and July (Witherington and Ehrhart 1989). During the nesting season, females nest at approximately 2-week intervals, laying an average of 3-4 clutches (Johnson and Ehrhart 1996). Clutch size often varies among subpopulations, but mean clutch size is around 110-115 eggs. In Florida, green sea turtle nests contain an average of 136 eggs (Witherington and Ehrhart 1989). Eggs incubate for approximately 2 months before hatching. Survivorship at any particular nesting site is greatly influenced by the level of anthropogenic stressors, with the more pristine and less disturbed nesting sites (e.g., along the Great Barrier Reef in Australia) showing higher survivorship values than nesting sites known to be highly disturbed [e.g., Nicaragua (Campbell and Lagueux 2005; Chaloupka and Limpus 2005)].

After emerging from the nest, hatchlings swim to offshore areas and go through a post-hatchling pelagic stage where they are believed to live for several years. During this life stage, green sea turtles feed close to the surface on a variety of marine algae and other life associated with drift lines and debris. This early oceanic phase remains one of the most poorly understood aspects of green sea turtle life history (NMFS and USFWS 2007b). Green sea turtles exhibit particularly slow growth rates of about 1-5 cm per year (Green 1993; McDonald-Dutton and Dutton 1998), which may be attributed to their largely herbivorous, low-net energy diet (Bjorndal 1982). At approximately 20-25 cm carapace length, juveniles leave the pelagic environment and enter nearshore developmental habitats such as protected lagoons and open coastal areas rich in sea grass and marine algae. Growth studies using skeletochronology indicate that green sea turtles in the western Atlantic shift from the oceanic phase to nearshore developmental habitats after approximately 5-6 years (Bresette et al. 2006; Zug and Glor 1998). Within the developmental habitats, juveniles begin the switch to a more herbivorous diet, and by adulthood feed almost exclusively on seagrasses and algae (Rebel and Ingle 1974). However, some populations are known to also feed heavily on invertebrates (Carballo et al. 2002). Green sea turtles reach sexual maturity at 20-50 years of age (Chaloupka and Musick 1997; Hirth and USFWS 1997), which is considered one of the longest ages to maturity of any sea turtle species.

While in coastal habitats, green sea turtles exhibit site fidelity to specific foraging and nesting grounds, and it is clear they are capable of "homing in" on these sites if displaced (McMichael et al. 2003). Reproductive migrations of Florida green sea turtles have been identified through flipper tagging and/or satellite telemetry. Based on these studies, the majority of adult female Florida green sea turtles are believed to reside in nearshore foraging areas throughout the Florida Keys and in the waters southwest of Cape Sable, with some post-nesting turtles also residing in Bahamian waters as well (NMFS and USFWS 2007b).

Status and Population Dynamics

Population estimates for marine turtles do not exist because of the difficulty in sampling turtles over their geographic ranges and within their marine environments. However, researchers have used nesting data to study trends in reproducing sea turtles over time. A summary of nesting trends is provided in the most recent 5-year status review for the species (NMFS and USFWS 2007b) organized by ocean region (i.e., Western Atlantic Ocean, Central Atlantic Ocean, Eastern Atlantic Ocean, Mediterranean Sea, Western Indian Ocean, Northern Indian Ocean, Eastern Indian Ocean, Southeast Asia, Western Pacific Ocean, Central Pacific Ocean, and Eastern Pacific Ocean). Trends at 23 of the 46 nesting beach sites reviewed in the 5-year status review found that nesting at 10 of the sites appeared to be increasing, nesting at 9 appeared to be stable, and nesting at 4 appeared to be decreasing. With respect to regional trends, the Pacific, the Western Atlantic, and the Central Atlantic regions appeared to show more positive trends (i.e., more nesting sites increasing than decreasing) while the Southeast Asia, Eastern Indian Ocean, and possibly the Mediterranean Sea regions appeared to show more negative trends (i.e., more nesting sites decreasing than increasing). These regional determinations should be viewed with caution since trend data was only available for about half of the total nesting concentration sites examined in the review and site-specific data availability appeared to vary across all regions.

The Western Atlantic region (i.e., the focus of this Opinion) was one of the best performing in terms of abundance in the entire review as there were no sites that appeared to be decreasing. The 5-year status review for the species identified 8 geographic areas considered to be primary sites for green sea turtle nesting in the Atlantic/Caribbean and reviewed the trend in nest count data for each (NMFS and USFWS 2007a). These sites include (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Achipelago, Guinea-Bissau. Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precluded a meaningful trend assessment for either (NMFS and USFWS 2007a). Seminoff (2004) likewise reviewed green sea turtle nesting data for 8 sites in the western, eastern, and central Atlantic, including all of the above with the exception that nesting in Florida was reviewed in place of Isla Trindade, Brazil. Seminoff (2004) concluded that all sites in the central and western Atlantic showed increased nesting, with the exception of nesting at Aves Island, Venezuela, where nesting is stable, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic; however, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007a). More information about site-specific trends for the other major ocean regions can be found in the most recent 5-year status review for the species (see NMFS and USFWS (2007a)).

By far, the largest known nesting assemblage in the Western Atlantic region occurs at Tortuguero, Costa Rica. According to monitoring data on nest counts, as well as documented emergences (both nesting and non-nesting events [i.e., false crawls]), there appears to be an increasing trend in this nesting assemblage since monitoring began in the early 1970s. For instance, from 1971-1975 there were approximately 41,250 average annual emergences documented and this number increased to an average of 72,200 emergences from 1992-1996 (Bjorndal et al. 1999). Troëng and Rankin (Troëng and Rankin 2005) collected nest counts from 1999-2003 and also reported increasing trends in the population consistent with the earlier studies, with nest count data suggesting 17,402-37,290 nesting females per year (NMFS and USFWS 2007a). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more resulted in an estimate of the Tortuguero, Costa Rica, population of nesting females growing at 4.9% annually.

In the continental United States, green sea turtle nesting occurs along the Atlantic coast, primarily along the central and southeast coast of Florida where an estimated 200-1,100 females nest each year (Meylan et al. 1994; Weishampel et al. 2003). Occasional nesting has also been documented along the Gulf coast of Florida (Meylan et al. 1995). More recently, green sea turtle nesting has occurred in North Carolina on Bald Head Island, just east of the mouth of the Cape Fear River, on Onslow Island, and on Cape Hatteras National Seashore. In 2010, a total of 18 nests were found in North Carolina, 6 nests in South Carolina, and 6 nests in Georgia (nesting databases maintained on www.seaturtle.org).

In Florida, FWRI has established index beaches to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the ten years of regular monitoring (Figure 7). According to data collected from Florida's index nesting beach surveys from 1989-2012, green sea turtle nest counts across Florida have increased approximately 10-fold from a low of 267 in the early 1990s to a high of 10,701 in 2011. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in both 2010 and 2011 followed by another decrease in 2012 (Figure 7). Modeling by Chaloupka et al. (2008) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%.

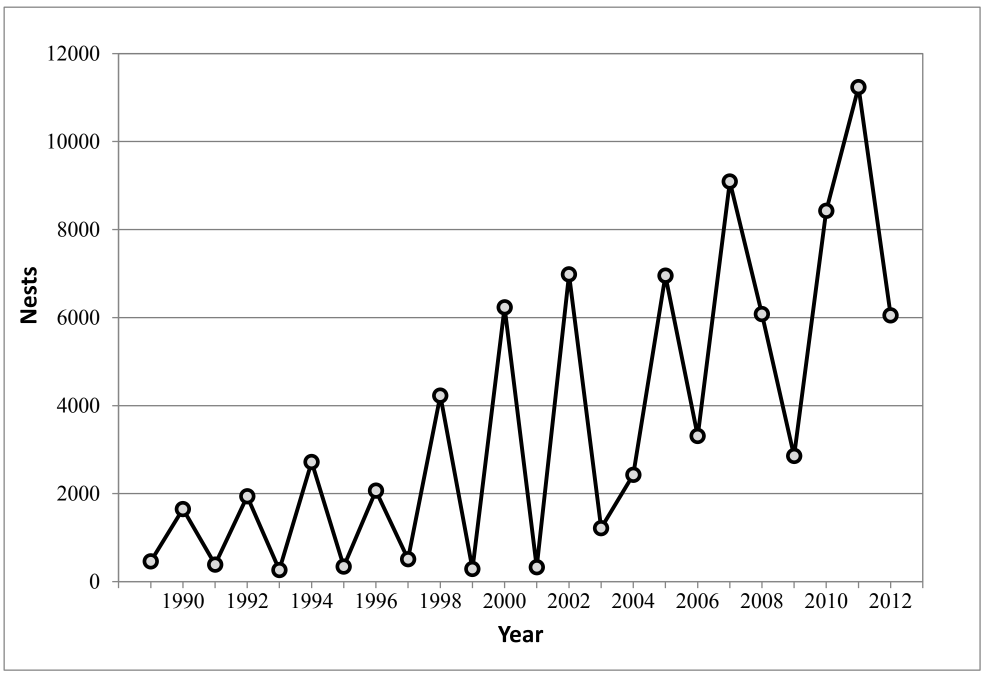

Figure 7. Green sea turtle nesting at Florida index beaches since 1989

Threats

The principal cause of past declines and extirpations of green sea turtle assemblages has been the overexploitation of the species for food and other products. Although intentional take of green sea turtles and their eggs is not extensive within the southeastern United States, green sea turtles that nest and forage in the region may spend large portions of their life history outside the region and outside U.S. jurisdiction, where exploitation is still a threat. Green sea turtles also face many of the same threats as other sea turtle species, including destruction of nesting habitat from storm events, oceanic events such as cold-stunning, pollution (e.g., plastics, petroleum products, petrochemicals, etc.), ecosystem alterations (e.g., nesting beach development, beach nourishment and shoreline stabilization, vegetation changes, etc.), poaching, global climate change, fisheries interactions, natural predation, and disease. A discussion on general sea turtle threats can be found in Section 4.2.1.1.

In addition to general threats, green sea turtles are susceptible to natural mortality from Fibropapillomatosis (FP) disease. FP results in the growth of tumors on soft external tissues (flippers, neck, tail, etc.), the carapace, the eyes, the mouth, and internal organs (gastrointestinal tract, heart, lungs, etc.) of turtles (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). These tumors range in size from 0.1 cm to greater than 30 cm in diameter and may affect swimming, vision, feeding, and organ function (Aguirre et al. 2002; Herbst 1994; Jacobson et al. 1989). Presently, scientists are unsure of the exact mechanism causing this disease, though it is believed to be related to both an infectious agent, such as a virus (Herbst et al. 1995), and environmental conditions [e.g., habitat degradation, pollution, low wave energy, and shallow water (Foley et al. 2005)]. Presently, FP is cosmopolitan, but has been found to affect large numbers of animals in specific areas, including Hawaii and Florida (Herbst 1994; Jacobson 1990; Jacobson et al. 1991).

Cold-stunning is another natural threat to green sea turtles. Although it is not considered a major source of mortality in most cases, as temperatures fall below 8°-10°C turtles may lose their ability to swim and dive, often floating to the surface. The rate of cooling that precipitates cold-stunning appears to be the primary threat, rather than the water temperature itself (Milton and Lutz 2003). Sea turtles that overwinter in inshore waters are most susceptible to cold-stunning because temperature changes are most rapid in shallow water (Witherington and Ehrhart 1989). During January 2010, an unusually large cold-stunning event in the southeastern United States resulted in around 4,600 sea turtles, mostly greens, found cold-stunned, with hundreds found dead or dying. A large cold-stunning event occurred in the western Gulf of Mexico in February 2011, resulting in approximately 1,650 green sea turtles being found cold-stunned in Texas. Of these, approximately 620 were found dead or died after stranding, and approximately 1,030 were rehabilitated and released. Additionally, during this same time frame, approximately 340 green sea turtles were found cold-stunned in Mexico, though approximately 300 of those were subsequently rehabilitated and released.

4.2.2 Corals: Staghorn, Mountainous Star, Knobby Star, Lobed Star, Elliptical Star, Lamarck's Sheet, and Rough Cactus

Elkhorn and staghorn corals were listed as threatened under the ESA in May 2006 (71 FR 26852). In December 2012, NMFS proposed changing their status from threatened to endangered (77 FR 73219); a final determination on the status change is still pending. Elkhorn coral does not occur within the project area.

Additionally, in December 2012, NMFS proposed to list 7 coral species (lobed star, mountainous star, knobby star, pillar, rough cactus, Lamarck's sheet, and elliptical star coral) in the western Atlantic, Gulf of Mexico, and/or Caribbean basins under the ESA. Five of those species are proposed as endangered, and 2 species are proposed as threatened (77 FR 73219; December 7, 2012). Pillar coral does not occur within the project area.

General information about corals that pertains to all the listed and proposed coral species is presented at the beginning of each of the subsections. Species-specific information is then presented for each of the listed and proposed coral species. However, lobed star, mountainous star, and knobby star corals are presented as a group since information is often only available for the species complex rather than the individual species.

Species Description
Corals are marine invertebrates in the phylum Cnidaria, which include true stony corals, the blue corals, and fire corals. All of the currently-listed and proposed corals in the NMFS Southeast Region (North Carolina through Texas and the U.S. Caribbean) are stony corals. Stony corals are characterized by polyps with multiples of 6 tentacles around the mouth for feeding and capturing prey items in the water column (Brainard et al. 2011a). Most stony corals form complex colonies made up of a tissue layer of polyps growing on top of a calcium carbonate skeleton, which the polyps produce through the process of calcification.

All of the listed and proposed-for-listing coral species are reef building species, which are capable of rapid calcification rates because of their symbiotic relationship with single-celled dinoflagellate algae, zooxanthellae, which occur in great numbers within the host coral tissues. Zooxanthellae photosynthesize during the daytime, producing an abundant source of energy for the host coral that enables rapid growth. At night, polyps extend their tentacles to filter-feed on microscopic particles in the water column, such as zooplankton, providing additional nutrients for the host coral. In this way, reef-building corals obtain nutrients autotrophically (i.e., via photosynthesis) during the day, and heterotrophically (i.e., via predation) at night (Brainard et al. 2011b).

Staghorn Coral
Staghorn coral (*Acropora cervicornis*; threatened, proposed endangered) coral (*Acropora palmata*; threatened, proposed endangered) are branching species that occur throughout the wider Caribbean. Staghorn corals have straight or slightly curved, cylindrical branches that look like deer antlers. The species range in color from golden yellow to brown, and the growing tips tend to be lighter or lack color. Individual staghorn coral colonies can reach up to 5 ft (1.5 m) across but may form thickets composed of multiple colonies that are difficult to tell apart. Staghorn corals are reef-building species that provide important habitat for other reef organisms, and other reef-building corals cannot fill the unique structural and ecological role of this coral species (Bruckner 2002a).

Lobed Star, Mountainous Star, and Knobby Star Corals
Lobed star coral (*Orbicella annularis*; proposed endangered), mountainous star coral (*Orbicella faveolata*; proposed endangered), and knobby star coral (*Orbicella franksi*; proposed endangered) are the 3 species in the *Orbicella annularis* complex. These 3 species were formerly in the genus *Montastraea*; however, recent work has reclassified the 3 species in the *annularis* complex to the genus *Orbicella* (Budd et al. 2012). The species complex was historically one of the primary reef framework builders throughout the wider Caribbean. The complex was considered a highly plastic, single species – *Montastraea annularis* – with growth forms ranging from columnar, to massive, to platy (formed of plates). In the early 1990s, Weil and Knowlton (1994) suggested the partitioning of these growth forms into separate species, resurrecting the previously described taxa, *Montastraea* (now *Orbicella*) *faveolata* and *Montastraea* (now *Orbicella*) *franksi*. These 3 sibling species were differentiated on the basis of

morphology, depth range, ecology, and behavior (Weil and Knowton 1994). Subsequent reproductive and genetic studies have generally supported the partitioning of the *annularis* complex into three species. *Orbicella faveolata* is the most genetically distinct while *Orbicella annularis* and *Orbicella franksi* are less so (Budd et al. 2012; Fukami et al. 2004; Lopez et al. 1999).

Some studies report on the species complex rather than individual species since visual distinction can be difficult where colony morphology cannot be discerned (e.g. small colonies or photographic methods). Information from these studies is reported for the species complex. Where species-specific information is available, it is reported. However, information about *O. annularis* published prior to 1994 will be attributed to the species complex since it is dated prior to the split of *O. annularis* into 3 separate species.

Lobed star coral colonies grow in columns that exhibit rapid and regular upward growth. Live colony surfaces usually lack ridges or bumps. Colonies can grow to several meters in height and diameter and are commonly grey, green, and brownish in color (Szmant et al. 1997).

Mountainous star corals grow in heads or sheets, the surface of which may be smooth or have keels or bumps. Colonies can reach up to 33 ft (10 m) in diameter with a height of 13-16 ft (4-5 m) and are commonly grey, green, and brownish in color (Szmant et al. 1997).

Knobby star corals are distinguished by large, unevenly arranged polyps that give the colony its characteristic irregular surface. Colony form is variable. Colonies can reach up to 16 ft (5 m) in diameter with a height of up to 6.5 ft (2 m) and are green, grey, and brown in color (Szmant et al. 1997).

Rough Cactus Coral
Rough cactus coral (*Mycetophyllia ferox*; proposed endangered) colonies are encrusting, flat plates. Colonies are thin, weakly attached plates with interconnecting, slightly sinuous, narrow valleys. Colonies are most commonly greys and browns in color with valleys and walls of contrasting colors, and their maximum size is 20 inches (50 cm) in diameter (Veron 2000).

Lamarck's Sheet Coral
Lamarck's sheet coral (*Agaricia lamarcki*; proposed threatened) forms flat or encrusting platy colonies that are commonly arranged in whorls. Colonies are brown in color, usually with pale margins. Polyp mouths are characteristically white and star-shaped. Maximum colony diameter is approximately 3 ft (1 m) (Veron 2000).

Elliptical Star Coral
Elliptical star coral (*Dichocoenia stokesii*; proposed threatened) colonies are either massive and spherical, or form thick, sub-massive plates. Although sometimes green, they are usually orange-brown with white margins between polyps (Veron 2000).

Distribution
In general, the corals in the Southeast Region are widely distributed throughout the western Atlantic, Caribbean, and Gulf of Mexico. Corals need hard substrate on which to settle and

form; however, only a narrow range of suitable environmental conditions allows coral to grow and exceed loss from physical, chemical, and biological erosion. Reef-building corals do not thrive outside a narrow temperature range of 25°C-30°C, but they are able to tolerate temperatures outside this range for brief periods of time, depending on how long and severe the exposure to extremes, as well as other biological and environmental factors. Two other important factors influencing suitability of habitat are light and water quality. Reef-building corals require light for photosynthesis of their symbiotic algae, and poor water quality can negatively affect both coral growth and recruitment. Availability of light generally limits how deep corals are found. Hydrodynamic condition (e.g., high wave action) is another important habitat feature, as it influences the growth, mortality, and reproductive rate of each species adapted to a specific hydrodynamic zone.

Staghorn Coral

Staghorn coral commonly grows in water ranging from 15 to 65 ft (5-20 m) in depth and rarely in waters to 196 ft (60 m) (Davis 1982; Jaap 1984; Jaap et al. 1989; Wells 1933). Staghorn coral is widely distributed throughout the western Atlantic and Caribbean. Areas occupied by this coral within U.S. jurisdiction are limited to 4 counties in the state of Florida, Puerto Rico, U.S. Virgin Islands, and Navassa Island. There is currently no evidence of range constriction for this species, though populations throughout the range have decreased substantially since the 1970s.

In Florida, staghorn coral has been documented along the east coast as far north as Palm Beach County. It occurs in deeper water (50-100 ft/16-30 m) at its northernmost range (Goldberg 1973; E. Tichenor, Palm Beach County Reef Rescue, pers. comm. to Jennifer Moore, NMFS 2008) and is distributed across its depth range (15-100 ft/5-30 m) off Broward and Miami-Dade Counties, the Florida Keys, and the Dry Tortugas (Jaap 1984). Off the shore of Broward County, staghorn corals form extensive thickets, which are the largest known remaining populations within U.S. jurisdiction. In Puerto Rico, coral reefs with varying densities of staghorn corals are off all coasts of the main island and around some of its smaller islands. Dense, tall thickets of staghorn coral are present in only a few reefs along the southwest, north, and west shore of the main island and isolated offshore locations (Schärer et al. 2009; Weil et al. 2002). In the U.S. Virgin Islands staghorn corals occur off St. Croix, St. Thomas, and St. John (Brainard et al. 2011a).

Lobed Star, Mountainous Star, and Knobby Star Corals

The 3 species in the *Orbicella annularis* complex (composed of lobed star coral (*Orbicella annularis*), mountainous star coral (*Orbicella faveolata*), and knobby star coral (*Orbicella franksi*) is distributed throughout the Caribbean, Bahamas, and Flower Garden Banks (IUCN 2010; Veron 2000). The complex occurs commonly throughout U.S. waters of the western Atlantic and Caribbean, including Florida (Martin though Monroe counties) and the Gulf of Mexico. The species occupy most reef environments, occurring in both protected and wave exposed habitats (Goreau and Wells 1967; Van Duyl 1985). Lobed star coral occurs shallower than its siblings, in depths ranging from 1.5-66 ft (0.5-20 m) (Szmant et al. 1997). Mountainous and knobby star corals can be found in depths up to 230 ft (70 m [Brainard et al. 2011a]).

Rough Cactus Coral

Rough cactus coral occurs throughout the U.S. waters of the western Atlantic, Caribbean, and Gulf of Mexico (Veron 2000), but has not been reported from Flower Garden Banks (Hickerson et al. 2008). It has also been observed in the Bahamas, but it is absent in the waters of Bermuda. The species occurs in shallow reef environments in depths ranging from 16-98 ft (5 to 30 m [Brainard et al. 2011a]).

Lamarck's Sheet Coral
Lamarck's sheet coral is distributed in the western Atlantic and throughout the Caribbean but is not known to occur in Bermuda (IUCN 2010). In U.S. waters, the species occurs in Florida (Goldberg 1973), Puerto Rico (Acevedo et al. 1989; Garcia-Sais 2010; Morelock et al. 2001), the Virgin Islands (Rogers et al. 1984; Smith et al. 2010), and Flower Garden Banks (Caldow et al. 2009). The species occurs in water depths ranging from 10-249 ft (3-76 m [Carpenter et al. 2008; Ghiold and Smith 1990; Humann 1993]). Although the species can rarely inhabit shaded areas in shallow waters, it primarily occurs at deeper depths. It also inhabits reef slopes and walls and can be one of the most abundant corals on deep reefs (Humann 1993).

Elliptical Star Coral
Elliptical star coral is distributed in the western Atlantic and throughout the Caribbean, the Gulf of Mexico, Florida (including the Florida Middle Grounds), the Bahamas, and Bermuda (Aronson et al. 2008). It is found in most reef environments within its range (Veron 2000), including both back reef and fore reef environments, rocky reefs, lagoons, spur-and-groove formations, channels, and occasionally at the base of reefs (Aronson et al. 2008). The species has been reported in water depths ranging from 6.5-236 ft (2-72 m) (Carpenter et al. 2008).

Life History Information
Corals use a number of diverse reproductive modes (Figure 8). Most coral species reproduce sexually and asexually. Corals reproduce sexually by developing eggs and sperm within the polyps. Some coral species have separate sexes (gonochoric), while others are both sexes at the same time (hermaphroditic). Strategies for fertilization are by "brooding" or "broadcast spawning" (i.e., internal or external fertilization, respectively). Asexual reproduction occurs through fragmentation when pieces of a colony break off and re-attach to hard substrate to form a new colony. Fragmentation results in multiple genetically-identical colonies. In many species of branching corals, fragmentation is a common and sometimes dominant means of propagation.

Depending on the mode of fertilization, coral larvae (called planulae) undergo development either mostly within the mother colony (brooders) or outside in the ocean (broadcast spawners). In either mode of larval development, planula larvae presumably experience considerable mortality (up to 90% or more) from predation or other factors prior to settlement and metamorphosis. Such mortality cannot be directly observed, but is inferred from the large amount of eggs and sperm spawned versus the much smaller number of recruits observed later. Coral larvae are relatively poor swimmers; therefore, their dispersal distances largely depend on how long they remain in the water column and the speed and direction of water currents transporting the larvae. The documented maximum larval life span is 244 days (*Montastraea magnistellata* [Graham et al. 2008]), which suggests that the potential for long-term dispersal of coral larvae, at least for some species, may be substantially greater than previously thought and may partially explain the large geographic ranges of many species.

Biological and physical factors that have been shown to affect spatial and temporal patterns of coral recruitment include:

- substratum availability and community structure (Birkeland 1977)
- grazing pressure (Rogers et al. 1984; Sammarco 1985)
- fecundity, mode, and timing of reproduction (Harriott 1985; Richmond and Hunter 1990)
- behavior of larvae (Goreau et al. 1981; Lewis 1974)
- hurricane disturbance (Hughes and Jackson 1985)
- physical oceanography (Baggett and Bright 1985; Fisk and Harriott 1990)
- the structure of established coral assemblages (Harriott 1985; Lewis 1974)
- chemical cues (Morse et al. 1988)

In general, upon proper stimulation coral larvae settle on appropriate substrates. Some evidence indicates that chemical cues from crustose coralline algae (CCA), microbial films, and/or other reef organisms (Gleason et al. 2009; Morse et al. 1996; Morse et al. 1994; Negri et al. 2001) or acoustic cues from fish and crustaceans in reef environments (Vermeij et al. 2010) stimulate settlement behaviors. Once a settlement site is chosen, the larvae attach to the surface and lay down a calcium carbonate skeleton. Successful recruitment of larvae is the only way new genetic individuals enter a population, thereby maintaining or increasing genotypic diversity (i.e., number of individuals if a population of clonal organisms). The larval stage is also important, as it is the only phase in the life cycle of corals where dispersal occurs over long distances. This helps genetically link populations and provides the potential to re-populate depleted areas. Because newly settled corals barely protrude above the substrate, juveniles need to reach a certain size to limit damage or mortality from threats such as grazing, sediment burial, and algal overgrowth (Bak and Elgershuizen 1976; Birkeland 1977; Sammarco 1985). Once recruits reach about 1-2 years post-settlement, growth and mortality rates appear similar across species. In some species, it appears that there is virtually no limit to colony size beyond structural integrity of the colony skeleton, as polyps apparently can bud indefinitely.

Stony corals require hard substrate for settlement of their larvae, and presence of other benthic organisms (e.g., macroalgae) can preclude settlement. Encrusting sponges and soft corals, zoanthids, and macroalgae are major coral competitors because of their ability to blanket large areas of the sea floor. The presence of macroalgae inhibits coral settlement both by competing for space and by trapping sediment that can abrade and smother small recruits. Juvenile corals are the most susceptible to overgrowth and mortality from these competitors, and corals are generally better able to compete as they grow larger (Bak and Elgershuizen 1976; Birkeland 1977).

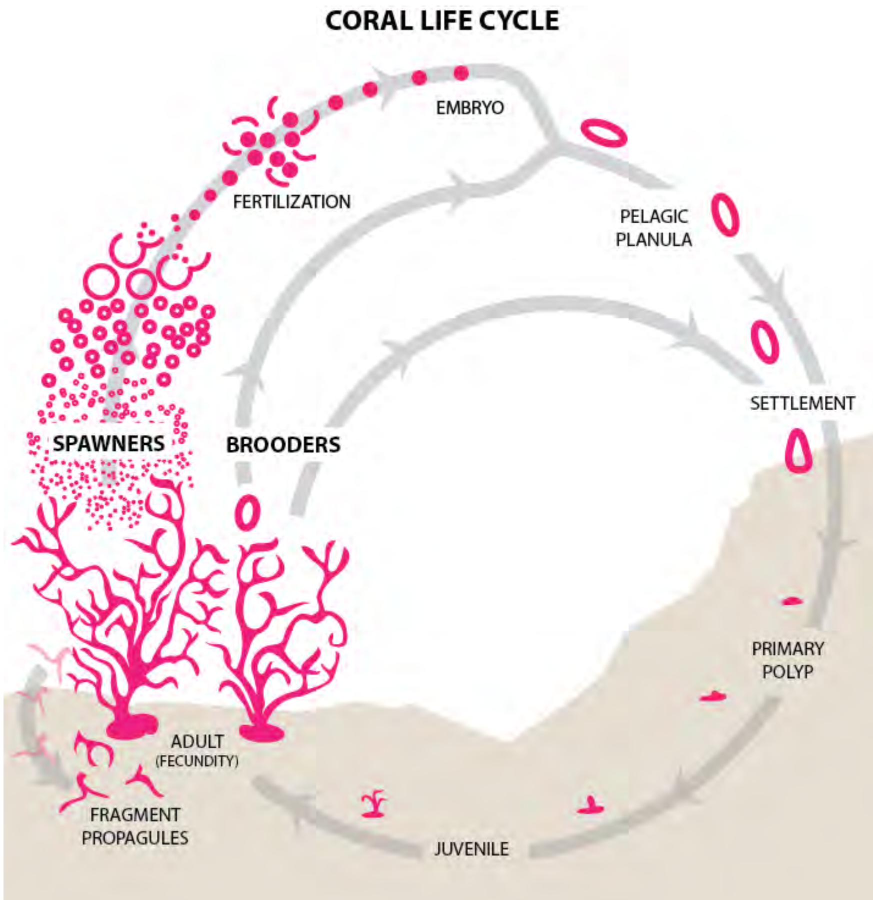

Figure 8. Coral life cycle showing different life history stages for broadcast spawners versus brooders, as well as asexual fragmentation (Reproduced from Brainard et al. 2011. Diagram prepared by Amanda Toperoff, NOAA PIFSC)

Staghorn Coral

Staghorn corals reproduce both sexually and asexually. Staghorn corals are hermaphroditic and are broadcast spawners (Szmant 1986). However, the species cannot self-fertilize, and 2 genetically distinct parents are required to produce viable larvae (Baums et al. 2005). Staghorn corals release gametes a few nights after the full moon during July, August, or September; however, some populations may have spawning events during 2 months. Staghorn colonies reach sexual maturity at 6.5 inches (17 cm) in branch length, but reproductive colonies 3.5 inches (9 cm) in branch length have been observed (Soong and Lang 1992). Skeletal growth rates are fast relative to other Caribbean coral species. Linear extension rates range from 1-4.5 inches (3-11.5 cm) per year for staghorn coral (Becker and Mueller 2001; Gladfelter et al. 1978; Jaap 1974; Shinn 1966; Shinn 1976; Vaughan 1915). New recruits and juveniles typically grow at slower rates. Larger colonies have higher fertility rates and produce proportionally more gametes than small colonies since basal and branch tip tissue are not fertile (Soong and Lang 1992). Fertilized eggs develop into planula larvae over several days in the water column. When larvae are ready to settle, they swim down to the bottom where they crawl along the surface searching for an appropriate settlement site. Certain species of CCA help settlement and post-settlement survival in staghorn coral (Ritson-Williams et al. 2009).

Lobed Star, Mountainous Star, and Knobby Star Corals

All 3 species of the *Orbicella annularis* complex are hermaphroditic broadcast spawners, with spawning concentrated on nights 6-8 following the full moon in late summer (Levitan et al. 2004). Fertilization success measured in the field was generally below 15 % for all 3 species but was highly linked to the number of colonies observed spawning at the same time (Levitan et al. 2004). Minimum size for reproduction of the *O. annularis* species complex was found to be 13 in^2 (83 cm^2) in Puerto Rico and was estimated to correspond to 4-5 years of age (Szmant-Froelich 1985). The *Orbicella annularis* species complex typically exhibits a linear growth of ~0.4 inches (1 cm) per year (Gladfelter et al. 1978), but increased appreciation for the slow rate of growth of post-settlement stages suggest this age for minimum reproductive size may be an underestimate (M.W. Miller, Southeast Fisheries Science Center, Miami, FL. pers. obs., October 2010). Growth rates of the *O. annularis* species complex are also negatively correlated with depth and water clarity (Hubbard and Scaturo 1985). The slow post-settlement growth rates of *O. faveolata* (Szmant and Miller 2005) and small eggs (Szmant et al. 1997) and larvae of all 3 species are factors that may contribute to extremely low post-settlement survivorship, even lower than other Caribbean broadcasters, such as elkhorn coral (Szmant and Miller 2005). Spatial distribution may also affect fecundity on the reef, with deeper colonies of *O. faveolata* being less fecund due to polyp spacing (Villinski 2003).

Successful recruitment by *Orbicella annularis* complex species has seemingly always been rare. (Hughes and Tanner 2000) reported the occurrence of only a single recruit of *Orbicella* over 18 years of intensive observation of 129 ft^2 (12 m^2) of reef in Discovery Bay, Jamaica, while many other recruitment studies throughout the Caribbean also report the species complex to be negligible to absent (Bak and Engel 1979; Rogers et al. 1984). *Orbicella* spp. juveniles also have higher mortality rates than larger colonies (Smith and Aronson 2006). Despite their generally boulder-like form, at least the lobed star coral is capable of some degree of fragmentation/fission and clonal reproduction (Foster et al. 2007).

Rough Cactus Coral

Rough cactus coral is a hermaphroditic brooder and polyps produce 96 eggs per cycle on average (Szmant 1986). It does not reproduce via fragmentation. Their larvae contain zooxanthellae (i.e., symbiotic algae) that can supplement maternal provisioning with energy sources provided by their photosynthesis (Baird et al. 2009). Colony size at first reproduction is greater than 15.5 in^2 (100 cm^2 [Szmant 1986]). Recruitment of this species appears to be very low; even studies from the 1970s reported zero settlement (Dustan 1977).

Lamarck's Sheet Coral

The specific reproductive strategy of Lamarck's sheet coral is presently unknown, but its congeners are primarily gonochoric brooders (i.e., separate sex individuals who partially rear larvae prior to release) (Delvoye 1988; Van Moorsel 1983). The larvae have been reported to primarily settle in relatively deep water (85-121 ft [26-37 m]), although the species has been found in shallow water (Bak and Engel 1979). Larvae of species within the genus are known to use chemical cues from CCA to indicate appropriate settlement substrate (Morse et al. 1988). The species has low recruitment rates. As an example, only one of 1,074 *Agaricia* recruits in a survey at the Flower Garden Banks may have been Lamarck's sheet coral (Shearer and Coffroth 2006). Net sexual recruitment over a decade can be negligible, with reproduction primarily via fragmentation (Hughes and Jackson 1985). Maximum size for Lamarck's sheet coral is up to ~6.5 ft (2 m) in diameter (Humann 1993), with radial growth rates in Jamaica ranging from 0-0.5 inches (0-1.4 cm) per year, but growing a bit more slowly in depths greater than 65 ft (20 m [Hughes and Jackson 1985]). Rogers et al. (1984) and Bak and Luckhurst (1980) have described the overall life history characteristics of Lamarck's sheet coral as roughly parallel to *Orbicella annularis*, that is, low overall recruitment rates, high survival, and high partial mortality. However, in Jamaica, Lamarck's sheet coral had faster growth, higher recruitment, and lower mortality rates than lobed star coral at the same site and depth (Hughes and Jackson 1985).

Elliptical Star Coral

Reproductive characteristics of elliptical star coral have been described from a histological study of populations in southeast Florida (Hoke 2007). This species is predominantly a gonochoric spawner with an overall sex ratio of 2:1 (male:female), but a small portion of hermaphroditic colonies (~18 %) were also observed in this southeast Florida population. Due to its morphology, elliptical star coral does not reproduce via fragmentation. Minimum colony size at reproduction was 25 in^2 (160 cm^2) in this population, and 2 potential spawning events per year were inferred: one in late August/early September and a second in October. Juvenile density has been reported as very low (Bak and Engel 1979) to relatively common in certain habitats (Chiappone 2010). The annual growth rate has been reported as 0.08-0.3 inches (2-7 mm) per year in diameter and 0.08-0.2 inches (2-5.2 mm) per year in height (Vaughn 1915).

Population Dynamics and Status

Documenting population dynamics for corals is confounded by several unique life history characteristics. Particularly, clonality and asexual reproduction makes it particularly difficult to census a species to determine population abundance estimates. This can only truly be done by tracking genotypically individual colonies within a set area over time to determine if a new colonies in the population are new sexual recruits or colonies formed by asexual reproduction or partial mortality (Williams et al. 2006). This is why coral abundance estimates are usually reported in percent cover rather than number of individuals.

Asexual reproduction can play a major role in maintaining local populations, but in the absence of sexual recruitment, it can also lead to decreased resilience to stressors due to decreased genetic diversity. Since corals cannot move and are dependent upon external fertilization to produce larvae, fertilization success declines greatly as adult density declines. In populations where fragmentation happens often, the number of genetically distinct adults is even lower than colony density. Likewise, when there are fewer adult colonies, there are also fewer sources of fragments to provide for asexual recruitment. These conditions imply that once a population declines to or below a certain level (i.e., the number of adults in an area is too low for sexual reproduction to be effective), the chances for recovery are low. Thus, local (reef-scale) reductions in colony numbers and size may prevent recovery for decades.

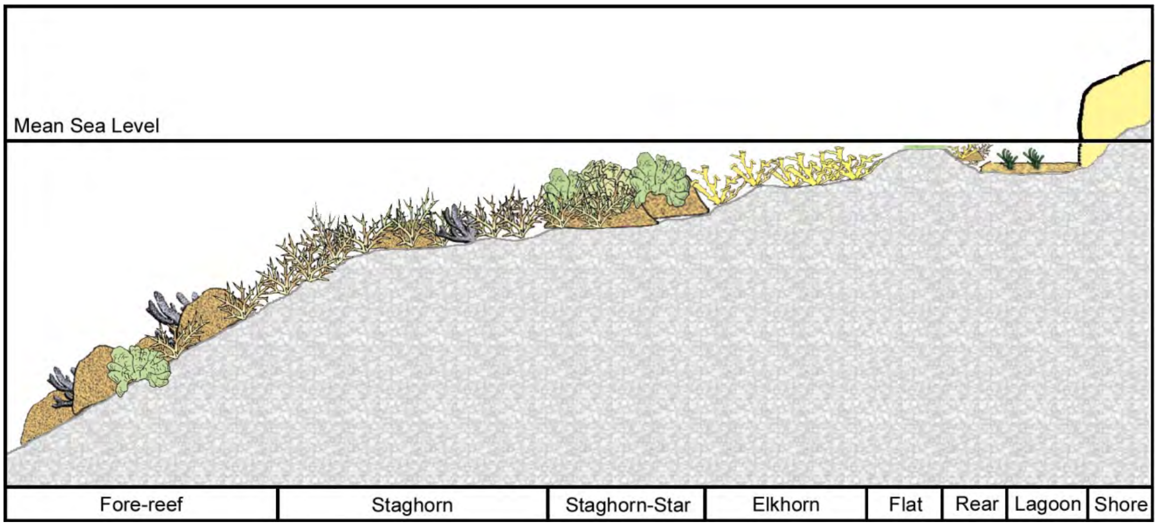

Figure 9. Generalized reef zone schematic (Acropora Biological Review Team 2005)

Staghorn Coral

Historically, staghorn coral was one of the dominant coral species and principle contributors to reef creation in the Atlantic and Caribbean. It commonly formed vast thickets, lending its names to a distinct zone in classical descriptions of Caribbean reef morphology (Figure 9). In the decades of the 1960s and 1970s, many Caribbean reefs were described as having an elkhorn (*A. palmata*) zone based upon high coverage, colony density, and in some cases, near exclusiveness of these species at particular depths (Goreau 1959).

Few historical estimates for staghorn coral population sizes are available because of its historically abundant status, its ability to produce clones through fragmentation, and its tendency to grow together to form complex thickets where individual colonies are difficult to tell apart. Although quantitative data on former distribution and abundance are scarce, in the few locations where quantitative data are available (e.g., Florida Keys, Dry Tortugas, Belize, Jamaica, and the U.S. Virgin Islands), declines in abundance (coverage and colony numbers) are estimated at greater than 97% (*Acropora* Biological Review Team 2005). Staghorn coral underwent precipitous declines throughout its range in the early 1980s due to mortality events associated with white band disease outbreaks and subsequent hurricane damage (Kramer 2002; Rogers et al. 2002). However, there are some small pockets of remnant robust populations such as in southeast Florida (Vargas-Angel et al. 2003), Honduras (Keck et al. 2005; Riegl et al. 2009), and the Dominican Republic (Lirman et al. 2010).

Miller et al. (2013) extrapolated population abundance of staghorn coral in the Florida Keys and Dry Tortugas from stratified random samples across habitat types. Population estimates of staghorn coral in the Florida Keys were 10.2 ± 4.6 (SE) million colonies in 2005, 6.9 ± 2.4 (SE) million colonies in 2007, and 10.0 ± 3.1 (SE) million colonies in 2012. In the Dry Tortugas, population estimates were 0.4 ± 0.4 (SE) million colonies in 2006 and 3.5 ± 2.9 (SE) million colonies in 2008, though the authors note their sampling scheme in the Dry Tortugas was not optimized for staghorn coral. In both the Florida Keys and Dry Tortugas, most of the population was dominated by small colonies less than 30 cm diameter. In the Florida Keys, partial mortality was highest in 2005, with up to 80% mortality observed, and lowest in 2007 with a maximum of 30%. In 2012, partial mortality ranged from 20%-50% across most size classes.

The recent trends in abundance for staghorn coral seem to conform to a pattern of stability punctuated by episodic, catastrophic declines. After the initial declines in the 1980s due to hurricanes and disease, a major El Niño/La Niña Southern Oscillation cycle in 1997-1998 resulted in a large bleaching event and a loss of coral in the Caribbean and the Atlantic (Wilkinson and Souter 2008).

There have been several reports of local trends in abundance. Lidz and Zawada (2013) observed 400 colonies of staghorn coral along 70.2 km of transects near Pulaski Shoal in the Dry Tortugas where the species had not been seen since the cold water die-off of the 1970s, but no thickets were observed. Cover of staghorn coral increased on a Jamaican reef from 0.6% in 1995 to 10.5% in 2004 (Idjadi et al.) and 44% by 2005, but then collapsed after the 2005 bleaching event and subsequent predation to less than 0.5% in 2006 (Quinn and Kojis 2008). Walker et al. (2012b) report increasing size of 2 thickets (expansion of up to 7.5 times the original size of 1 of the thickets) monitored off southeast Florida, but also noted that cover within monitored plots concurrently decreased by about 50%, highlighting the dynamic nature of staghorn coral as it moves around via fragmentation and re-attachment.

Riegl et al. (2009) monitored staghorn coral in photo plots on the fringing reef near Roatan, Honduras from 1996 to 2005. Staghorn coral cover was 0.42% in 1996, declined to 0.14% in 1999 after the Caribbean bleaching event in 1998 and mortality from runoff associated with a Category 5 hurricane, and decreased further to 0.09 % in 2005. Staghorn coral colony frequency decreased 71% between 1997 and 1999. In sharp contrast, offshore banks near Roatan had dense thickets of staghorn coral with 31% cover in photo-quadrats in 2005 and appeared to survive the 1998 bleaching event and hurricane, most likely due to bathymetric separation from land and greater flushing. Modeling showed that under undisturbed conditions, retention of the dense staghorn coral stands on the banks off Roatan is likely with a possible increased shift towards dominance by other coral species. (Riegl et al. 2009).

A report on the status and trends of Caribbean corals over the last century indicates that after the large mortality events of the 1970s and 1980s, cover of staghorn coral has remained relatively stable (though much reduced) throughout the region as has the frequency of reefs at which staghorn coral was described as the dominant coral (IUCN 2013). However, the report also indicates that the number of reefs with staghorn coral present declined during the 1980s, remained relatively stable (though lower) in the 1990s, and then continued to decrease through 2011.

Fragmentation is the most common way of forming new colonies in staghorn corals (Bak and Criens 1982; Davis 1977; Gilmore and Hall 1976; Hughes 1985; Tunnicliffe 1981). However, staghorn coral retains moderate to high levels of genotypic diversity (i.e., the ratio of genetically distinct individuals to all colonies in a population) in many geographic areas (Baums et al. 2010; Baums et al. 2006; Vollmer and Palumbi 2007), though areas with low levels of genotypic diversity also exist. Baums et al. (2010) report staghorn coral at other Florida sites showed higher levels of diversity, indicating a more even reliance on sexual and asexual reproduction.

Studies have found that genetic exchange is restricted between populations separated by greater than 300 miles (500 km), emphasizing the importance of locally diverse populations for the recovery of this species (Baums et al. 2010; Baums et al. 2006; Vollmer and Palumbi 2007).

Settlement of staghorn larvae is rarely detected in coral recruitment studies (Bak and Engel 1979; Rylaarsdam 1983; Sammarco 1980). Studies from across the wider Caribbean, however, confirm 2 overall patterns of sexual recruitment of staghorn corals: (1) low juvenile densities relative to other coral species; and (2) low juvenile densities relative to the commonness of adults (Porter 1987). This pattern suggests that the composition of the adult population is dependent upon variable recruitment and likely reflects the dominance of asexual reproduction by fragmentation for these species (i.e., surviving fragments are usually large and never undergo a "juvenile" stage). Fragmentation can provide a mechanism for locally maintaining and expanding staghorn coral populations. In many locations, populations of staghorn coral have been reduced to such an extent that the potential for recovery through re-growth of fragments is limited. Similarly, as the density of staghorn coral colonies has declined, gametes become diluted, and successful sexual reproduction is less likely and results in reduced potential for exchange of genetic material between populations that are spatially farther apart (Bruckner 2002b). Contributing to density concerns for staghorn coral are observations that spawning does not occur at the same time. Observations at sites in the Florida Keys where distinct genotypes do co-occur in close proximity indicate that they often spawn on different nights preventing effective larval production (Miller et al. unpublished data). Thus, there is evidence to suggest that sexual recruitment of staghorn coral is currently compromised and limiting the potential for recovery.

Lobed Star, Mountainous Star, and Knobby Star Corals
As described above, the 3 species in the *Orbicella annularis* complex were not suggested for formal separation until the mid-1990s and further supported by genetic studies through 2012 (Budd et al. 2012; Fukami et al. 2004; Lopez et al. 1999; Weil and Knowton 1994). In addition, the three species are potentially difficult to tell apart depending on their growth form (e.g., mounding versus platy) and survey method (e.g., video versus in situ). Therefore, many monitoring programs continue to lump the 3 species into the *O. annularis* complex. Future, focused studies may allow for more time to do field identification resulting in high confidence that the reported species is actually the one identified.

The *Orbicella annularis* complex has historically been dominant on Caribbean coral reefs, characterizing the so-called "buttress zone" and "annularis zone" in the classical descriptions of Caribbean reefs (Goreau 1959). There is ample evidence that it has declined dramatically throughout its range, but perhaps at a slower pace than staghorn corals. While the latter began its rapid decline in the early- to mid-1980s, declines in *Orbicella annularis* complex have been much more obvious in the 1990s and 2000s, most often associated with combined disease and bleaching events. In most cases where examined, additional demographic changes accompany these instances of declining abundance (e.g., size structure of colonies, partial mortality).

In Florida, the percent cover data from 4 fixed sites have shown the *Orbicella annularis* complex declined in absolute cover from 5% to 2% in the Lower Keys between 1998 and 2003, and was accompanied by 5% to 40% colony shrinkage and virtually no recruitment (Smith et al. 2008).

Earlier studies from the Florida Keys indicated a 31% decline of *Orbicella annularis* complex absolute cover between 1975 and 1982 at Carysfort Reef (Dustan and Halas 1987) and greater than 75% decline (from over 6% cover to less than 1%) across several sites in Biscayne National Park between the late 1970s and 2000 (Dupont et al. 2008). Further, Ruzicka et al. (2013) documented a Florida Keys-wide decline in all stony coral cover attributable to a decline in the *O. annularis* complex from 1999 to 2009. Most notably, they documented a 25% decline at the deep fore reef sites, where declines are typically not as dramatic. Taken together, these data imply extreme declines in the Florida Keys (80%–95%) between the late 1970s and 2003, and it is clear that further dramatic losses occurred in this region during the cold weather event in January 2010 (Colella et al. 2012).

Similar declines have also been documented for relatively remote Caribbean reefs. At Navassa Island National Wildlife Refuge, percent cover of *Orbicella annularis* complex on randomly sampled patch reefs declined from 26% in 2002 to 3% in 2009, following disease and bleaching events in this uninhabited oceanic island (Miller and Williams 2007). Additionally, 2 offshore islands west of Puerto Rico (Mona and Desecheo) showed reductions in in *O. annularis* complex species (*O. faveolata* and *O. annularis)* live colony counts of 24% and 32% between 1998-2000 and 2008, respectively (Bruckner and Hill 2009). At Desecheo, this demographic decline of one-third of the population corresponded to a decline in *Orbicella annularis* complex cover from over 35% to below 5% across 4 sites.

In the U.S. Virgin Islands, recent data from the U.S. National Park Service's Inventory and Monitoring Program across 6 sites at fixed stations show a decline of *Orbicella annularis* complex from just over 10% cover in 2003 to just over 3% cover in 2009 following mass bleaching and disease impacts in 2005 (Miller et al. 2009). This degree of recent decline was preceded by a decline from over 30% *Orbicella* coverage to approximately 10% between 1988 and 2003, as documented by Edmunds and Elahi (2007). Similarly, percent cover of *Orbicella annularis* complex in a marine protected area in Puerto Rico declined from 49% to 8% between 1997 and 2009 (Hernández-Pacheco et al. 2011). Taken together, these data suggest an 80%-90% decline in *Orbicella annularis* over the past 2 decades in the main U.S. Caribbean territories.

While Bak and Luckhurst (1980) indicated stability in *Orbicella annularis* complex cover across depths in Curaçao during a 5-year study in the mid-1970s, this region has also manifested *Orbicella annularis* complex declines in recent years. Bruckner and Bruckner (2006) documented an 85% increase in the partial mortality of *Orbicella faveolata* and *O. annularis* colonies across 3 reefs in western Curaçao between 1998 and 2005, approximately twice the level for all other stony corals combined. These authors noted that *Orbicella franksi* fared substantially better than the other two complex species in this study. It is likely that *Orbicella annularis* complex populations in Curaçao have fared better than other Caribbean regions, but even those populations are not immune to losses.

Orbicella annularis complex declines in additional locations are noted. For example, at Glovers Reef, Belize, McClanahan and Muthiga (1998) documented a 38%-75% decline in relative cover of *Orbicella annularis* complex across different reef zones between 1975 and 1998, and a further 40% decline in relative cover has occurred since then (Huntington et al. 2011). In contrast, , *O.*

franksi, *O. faveolata*, and *O. annularis* populations have shown stable status at sites in Colombia between 1998 and 2003 (Rodriguez-Ramirez et al. 2010), although demographic changes in *Orbicella annularis* at both degraded and less-degraded reefs imply some degree of population decline in this region (Alvarado-Chacón and Acosta 2009).

Rough Cactus Coral

Rough cactus coral is usually uncommon (Veron 2000) or rare according to published and unpublished records. It constitutes less than 0.1% species contribution (percent of all colonies surveyed) and occurs at densities less than 0.08 colonies per 1 m^2 in Florida (Wagner et al. 2010) and at 0.8 colonies per 100 m transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (Ginsburg and Lang 2003). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera, Puerto Rico, and St. Croix (USVI/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show *Mycetophyllia ferox* cover to be consistently less than 1%, with occasional observations up to 2%, and no apparent temporal trend.

Dustan (1977) suggests that *Mycetophyllia ferox* was much more abundant in the upper Florida Keys in the early 1970s than current observations, but that it was highly affected by disease. This data could be interpreted as a substantial decline. Long-term Coral Reef Evaluation and Monitoring Project (CREMP) data in Florida on species presence/absence from fixed stations also show a dramatic decline. For 97 stations in the main Florida Keys, occurrence had declined from 20 stations in 1996 to 4 stations in 2009; in Dry Tortugas occurrence had declined from 8 out of 21 stations in 2004 to 3 stations in 2009 (R. Ruzicka and M. Colella, Florida Marine Research Institute, St. Petersburg, Florida pers. comm. to Jennifer Moore, NMFS, Oct 2010). Recruitment of this species appears to be very low; even studies from the 1970s reported zero settlement (Dustan 1977).

Lamarck's Sheet Coral

Lamarck's sheet coral has been reported to be common (Veron 2000). On reefs at 98-131 ft (30-40 m) depths in the Netherlands Antilles, *Agaricia lamarcki* increased (Bak and Nieuwland 1995) or showed no decline in abundance from 1973 to 1992 (Bak et al. 2005), even though other corals on the same deep reefs decreased. It is not known whether this relative stability at depth holds across the full range of the species. The species has low recruitment rates. As an example, only 1 of 1,074 *Agaricia* recruits in a survey at the Flower Garden Banks may have been *Agaricia lamarcki* (Shearer and Coffroth 2006). Sexual recruitment over a decade has been documented as negligible, with reproduction primarily via fission (Hughes and Jackson 1985). It is a relatively long-lived species (Hughes 1996), with some colonies living more than a century (Hughes and Jackson 1985).

Elliptical Star Coral

Elliptical star coral is usually uncommon (Veron 2000). The overall colony density of *Dichocoenia stokesi* averaged across all habitat types in the south Florida region was ~1.6 colonies per 10 m^2, making it the ninth most abundant coral species in this region (Wagner et al. 2010). Substantial population declines have been reported from a bay in Curaçao (80% decline between 1961 and 1992; Debrot et al. 1998) and the upper Florida Keys (mortality of 75% of colonies across several reef sites after a disease outbreak with no recovery after 7 years;

[Richardson and Voss 2005]). There have been no obvious trends in the abundance of *Dichocoenia stokesi* in monitoring of randomized stations at La Parguera, Puerto Rico or St. John and St. Croix, U.S. Virgin Islands with less than 1.5% cover at most sites. Bak and Engel (1979) reported very low densities of *Dichocoenia* juveniles (approximately 1% of total juvenile colonies). However, reports of juveniles of *Dichocoenia stokesi* have been relatively common compared to most other scleractinian corals in the Florida Keys with mean juvenile densities among 566 sites surveyed during 1999–2009 averaging 0.11 per m^2, but reaching densities as high as one juvenile per m^2 in certain habitats (Chiappone 2010).

Threats

Ocean Warming

Mean seawater temperatures in reef-building coral habitats have increased during the past few decades and are predicted to continue to rise between now and 2100 (IPCC 2013). More importantly, the frequency of warm-season temperature extremes (warming events) in reef-building coral habitat has increased during the past 2 decades and is also predicted to increase between now and 2100 (IPCC 2013). The primary observable coral response to ocean warming is bleaching of coral colonies, wherein corals expel their symbiotic algae (zooxanthellae) in response to stress. Bleaching can affect coral growth, maintenance, reproduction, and survival. An episodic increase of only 1°C-2°C above the normal local seasonal maximum ocean temperature can induce bleaching. Although corals can withstand mild to moderate bleaching, severe, repeated, or prolonged bleaching can lead to colony death and has led to the mass mortality of many coral species during the past 30 years.

In addition to coral bleaching, ocean warming detrimentally affects virtually every life-history stage in reef-building corals. For one Indo-Pacific *Acropora* species, abnormal embryonic development occurs at 32°C, and complete fertilization failure occurs at 34°C (Negri et al. 2007). Further, symbiosis establishment, larval survivorship, and settlement success are impaired in some coral species at temperatures as low as 30°C-32°C (Randall and Szmant 2009; Ross et al. 2013; Schnitzler et al. 2012). Warmer temperatures accelerate the rate of larval development for spawning species, which reduces dispersal distances, the likelihood of successful settlement, and the potential for replenishment of depleted areas (Randall and Szmant 2009).

Multiple threats stress corals simultaneously or sequentially, whether the effects are cumulative, synergistic, or antagonistic. Ocean warming is likely to interact with many other threats, especially considering the long-term consequences of repeated thermal stress, since ocean warming is expected to worsen over this century. Increased seawater temperature interacts with coral diseases to reduce coral health and survivorship. Coral disease outbreaks often have accompanied or immediately followed bleaching events and follow seasonal patterns of high seawater temperatures. The effects of greater ocean warming (i.e., increased bleaching, which kills or weakens colonies) are expected to interact with the effects of higher storm intensity (i.e., increased breakage of dead or weakened colonies) in the Caribbean, resulting in increased rates of coral declines. Likewise, land-based runoff, pollution, or other local stressors may worsen bleaching impacts by increasing coral susceptibility to bleaching and/or increasing the duration of lowered growth after a bleaching event (Carilli et al. 2009; Wooldridge 2009).

Ocean Acidification

Ocean acidification is a result of increased greenhouse gas accumulation, primarily carbon dioxide, in the atmosphere. Ocean acidification is a drop in the pH of seawater that occurs in response to increases in atmospheric carbon dioxide levels that change ocean carbonate chemistry (Caldeira and Wickett 2003). The aragonite saturation state measures the concentration of carbonate ions in the ocean. Corals use carbonate ions to build calcium carbonate skeletons. Thus, decreasing pH and aragonite saturation state are expected to have a major impact on corals and other marine organisms this century by making it more difficult for them to build their skeletons (Fabry 2008). Numerous laboratory and field experiments have shown a relationship between elevated carbon dioxide and decreased calcification rates in particular corals and other calcium carbonate secreting organisms such as CCA (Bates et al. 2009; De Putron et al. 2010; Doney et al. 2009; Langdon et al. 2003). Low-saturation-state water also decreases the rate of biochemical processes that create the cements that infill reefs. A major potential impact from ocean acidification is a reduction in the structural stability of corals and reefs, which results both from increases in bioerosion and decreases in reef cementation. As atmospheric carbon dioxide rises globally, reef-building corals are expected to calcify more slowly and become more fragile.

Laboratory experiments have shown that a declining aragonite saturation state slows the start of and the rate at which newly settled coral larvae create carbonate skeletons (Albright et al. 2008; Cohen et al. 2007; Cohen et al. 2009). Slower growth implies even higher rates of mortality for newly settled corals that are vulnerable to overgrowth competition, sediment smothering, and incidental predation until they reach a refuge at larger colony size. In addition to effects on growth and calcification, recent laboratory experiments have shown that increased carbon dioxide also substantially impairs coral fertilization and settlement success (Albright et al. 2010), suggesting a potential further reduction in recruitment. Community medium-scale studies (Jokiel et al. 2008; Kuffner et al. 2008) showed dramatic declines in the growth rate of CCA and other reef organisms and an increase in the growth of fleshy algae at atmospheric carbon dioxide levels expected later this century. The decrease in CCA growth, coupled with rapid growth of fleshy algae will result in less available habitat for settlement and recruitment of new coral colonies.

Acidification is likely to interact with other threats. Ocean acidification may reduce the temperature threshold at which bleaching occurs (Anthony et al. 2011). Reduced skeletal growth compromises the ability of coral colonies to compete for space against algae, which grows more quickly as nutrient over-enrichment increases. Reduced skeletal density weakens coral skeletons, resulting in greater colony breakage from natural and human-induced physical damage.

Disease

Coral diseases are common and significant threats affecting most coral species. Disease can cause mortality, reduced sexual and asexual reproductive success, and impaired colony growth. A diseased state results from a complex interplay of factors including the cause or agent (e.g., pathogen, environmental toxicant), the host, and the environment. In the case of corals, the host is a complex community of organisms, which includes the coral animal, symbiotic zooxanthellae, and microbial symbionts.

Scientific understanding of individual disease causes in corals remains very poor. Lack of identification of specific pathogens of many coral diseases has hindered the ecological understanding of diseases and the ability to manage them effectively. Several authors have suggested there is a link between increased incidence of coral disease with increased temperature (Bruno et al. 2007; Harvell et al. 1999; Muller et al. 2008; Patterson et al. 2002) that may make corals more prone to infection or make pathogens more potent. An increased prevalence of infectious disease outbreaks has been associated with thermal stress even at temperatures below those required to cause mass bleaching (Bruno et al. 2007). In addition, disease outbreaks have followed bleaching events (Brandt and McManus 2009) and hurricanes (Bruckner and Bruckner 1997; Halley et al. 2001; Miller and Williams 2007; Williams et al. 2008), indicating greater susceptibility to disease when corals are stressed.

Trophic Effects of Fishing

Fishing, particularly overfishing, can have large scale, long-term ecosystem-level effects that can change ecosystem structure from coral-dominated reefs to algal-dominated reefs called a 'phase shift' (Hughes 1994). Phase shifts can result when fishing removes species that are particularly important in structuring coral reef ecosystems (Mumby et al. 2007). Effects of fishing can include reducing population abundance of herbivorous fish species that control algal growth, limiting the size structure of fish populations, reducing species richness of herbivorous fish, and releasing corallivores from predator control. If herbivorous fish populations, particularly large-bodied parrotfish, are heavily fished and a major mortality of coral colonies occurs, then algae can grow rapidly and prevent the recovery of the coral population. The ecosystem may then collapse into an alternative stable state– a persistent phase shift in which algae replace corals as the dominant reef species (Mumby et al. 2007). Although algae can have negative effects on adult coral colonies (i.e., overgrowth, bleaching from toxic compounds), the ecosystem-level effects of algae are primarily from inhibited coral recruitment. Filamentous algae can prevent the recruitment of coral larvae by creating sediment traps that obstruct access to a hard substrate for attachment. Additionally, macroalgae reduces coral recruitment through occupation of the available space, shading, abrasion, chemical poisoning, and infection with bacterial disease (Rasher et al. 2012; Rasher and Hay 2010; Rasher et al. 2011).

The trophic effects of fishing are likely to interact with many other threats. For example, when carnivorous fishes are overfished, corallivorous fish populations may increase, resulting in greater predation on corals (Burkepile and Hay 2007). Further, some corallivores are vectors of disease and can transmit disease from one coral colony to another as they transit and consume from each coral colony (Aeby and Santavy 2006). Increasing corallivore abundance results in transmittal of disease to higher proportions of the corals within the population.

Sedimentation

Human activities in coastal watersheds introduce sediment into the ocean by a variety of mechanisms; including river discharge, surface run-off, groundwater seeps, and atmospheric deposition. Elevated sediment levels are generated by poor land use practices and coastal and nearshore construction, including dredging. Nearshore sediment levels will also likely increase with sea level rise due to erosion at the shoreline and re-suspension of lagoonal sediments.

The most common direct effect of sedimentation is deposition of sediment on coral surfaces as it settles out from the water column. Corals with certain morphologies (e.g., mounding) can passively reject settling sediments or corals can actively displace sediment by ciliary action or mucous production, both of which require energetic expenditures (Bak and Elgershuizen 1976; Dallmeyer et al. 1982; Lasker 1980; Stafford-Smith 1993; Stafford-Smith and Ormond 1992). Corals that are unsuccessful in removing sediment will be smothered and die (Golbuu et al. 2003; Riegl and Branch 1995; Rogers 1983). Sediment can also induce sublethal effects, such as reductions in tissue thickness (Flynn et al. 2006) and excess mucus production (Marszalek 1981). In addition, suspended sediment can reduce the amount of light in the water column, making less energy available for coral photosynthesis and growth (Anthony and Hoegh Guldberg 2003; Bak 1978; Rogers 1979). While some corals may be more tolerant of short-term elevated levels of sedimentation, sediment stress and turbidity can induce bleaching (Philipp and Fabricius 2003; Rogers 1979). Finally, sediment impedes fertilization of spawned gametes (Gilmour 2002; Humphrey et al. 2008) and reduces larval settlement, as well as the survival of recruits and juveniles (Birrell et al. 2005; Fabricius et al. 2003).

Sedimentation is also likely to interact with many other threats. For example, when coral communities that are chronically affected by sedimentation experience a warming-induced bleaching event and associated disease outbreaks, the consequences for corals can be much more severe than in communities not affected by sedimentation.

Nutrients

Nutrients (e.g., nitrogen and phosphorous) are added to coral reefs from both point sources (readily identifiable inputs from a single source such as a pipe or drain) and non-point sources (inputs that occur over a wide area and are associated with particular land uses). Anthropogenic sources of nutrients include sewage, agricultural run-off, river and inlet discharges, and groundwater. Development of coastlines and destruction of mangrove forests compound the problem of anthropogenic nutrient runoff, as mangroves are able to filter massive amounts of nutrients and sediment caused by development. Natural processes bring nutrients to coral reefs as well, such as delivery of nutrient-rich deep water by internal waves and upwelling.

Elevated nutrients affect corals through 2 main mechanisms: direct impacts on coral physiology and indirect effects through nutrient-stimulation of other community components (e.g., macroalgae and filter feeders) that compete with corals for space on the reef. Coral reefs are adapted to low nutrient levels, and overabundance of nutrients can cause an imbalance that affects the entire ecosystem. Nutrient-rich water can enhance benthic algae and phytoplankton growth rates in coastal areas, resulting in overgrowth, competition, and algal blooms. Excess nutrient loads affect coral physiology and the balance between corals and their zooxanthellae (Szmant 2002). Increased nutrients can decrease calcification and reduce skeletal density. Either condition results in corals that are more prone to breakage or erosion. Increased levels of nutrients can also compromise coral health (Hodel and Vargas-Angel 2007). Notably, individual species have varying tolerance to increased nutrients.

Nutrients are likely to interact with many other threats. For example, when coral communities that are chronically affected by nutrients experience a warming-induced bleaching event and associated disease outbreaks, the consequences for corals can be much more severe than in communities not affected by nutrients. Additionally, experimental studies on diseased coral species indicate that nutrient augmentation adjacent to active disease lesions substantially increases disease severity (Bruno et al. 2003).

Sea Level Rise

Sea level rise may affect various coral life history events, including larval settlement, polyp development, and juvenile growth. It may also contribute to adult mortality and colony fragmentation, mostly due to increased sedimentation and decreased water quality (reduced light availability) caused by coastal inundation. The best available information suggests that sea level will continue to rise due to thermal expansion and the melting of land and sea ice. Many corals that inhabit the relatively narrow zone near the ocean surface have rapid growth rates when healthy, which allowed them to keep up with sea-level rise during the past periods of rapid climate change associated with de-glaciation and warming. However, depending on the rate and amount of sea level rise, rapid rises can lead to reef drowning. Rapid rises in sea level could affect many coral species by both submerging them below their common depth range and, more likely, by degrading water quality through coastal erosion and potentially severe sedimentation or enlargement of lagoons and shelf areas.

Rising sea level is likely to cause mixed responses in coral species depending on their depth preferences, sedimentation tolerances, and growth rates. Further, the nearshore topography can affect the impact sea level rise has on corals. Reductions in growth rate due to local stressors, bleaching, infectious disease, and ocean acidification may prevent the species from keeping up with sea level rise (e.g., from growing at a rate that will allow them to continue to occupy their preferred depth range despite sea-level rise). Additionally, lack of suitable new habitat, limited success in sexual recruitment, coastal runoff, and transition from natural to constructed shorelines will compound some corals' ability to survive rapid sea level rise.

Predation

Predation on some coral genera, including *Acropora and Orbicella*, is a chronic, though occasionally acute, energy drain (Cole et al. 2008; Rotjan and Lewis 2008). Predators of Caribbean corals include snails, polychaete worms, and several species of fishes. The effects of chronic and frequent predation on corals are usually inconsequential but can become significant once the coral population decreases below a threshold. If the living coral cover is substantially reduced by natural or anthropogenic disturbances, the effects of predation become larger even if the rate of predation does not change. The increased focus of predation on the fewer remaining colonies causes the colony to use energy in defense and could result in a reduced rate of healing and/or fecundity or reduced resistance to stressors and/or disease. Additionally, corallivore populations can also increase due to removal of carnivorous predators (i.e., predators of the corallivores) through fishing. Over-predation can lead to significant coral declines when the rate of coral predation is higher than the rate of healing or coral population replenishment.

Predation is likely to interact with other threats. For instance, predation of coral colonies can increase the likelihood of coral disease infection, and likewise diseased colonies may be more likely to be preyed upon. Additionally, nutrient runoff from land stimulates phytoplankton blooms, which provide food for the larvae of invertebrate corallivores and can cause outbreaks of these predators (Birkeland 1982; Fabricius et al. 2010).

Toxins and Contaminants

Toxins and bioactive contaminants may be delivered to coral reefs via either point or non-point sources. The general effects of contaminants on coral communities are reductions in coral growth, coral cover, and coral species richness (Keller et al. 1991; Loya and Rinkevich 1980; Pait et al. 2007), and a shift in community composition to more tolerant species (Rachello-Dolmen and Cleary 2007). Contaminant effects are species specific and may have harmful effects in combination that would not be evident under experimental exposure to an individual substance.

Laboratory experiments have shown chemical contaminants are harmful to corals. However, linking coral decline to specific contaminants in the environment can be difficult. Low concentrations (parts per billion) of organic chemical contaminants including hydrocarbons (Negri and Heyward 2000), antifoulants (Knutson et al. 2012), pesticides (Negri and Heyward 2001), and metals such as copper, zinc, and iron (Bielmyer et al. 2010; Reichelt-Brushett and Harrison 2000; Reichelt-Brushett and Harrison 2005; Vijayavel et al. 2012) can impact physiological function at various life stages. Estrogen compounds at concentrations that occur in urban or sewage-affected coastal waters (i.e., 2 ng L^{-1}) can affect coral growth and fecundity (Tarrant et al. 2004). In lab experiments, various compounds found in common sunscreens caused coral bleaching (Danovaro et al. 2008). Both oil and chemical dispersants are toxic to coral larvae (Epstein et al. 2000; Negri and Heyward 2000; Goodbody-Gringley et al., unpublished data; K. Ritchie, Mote Marine Lab, pers. comm. to A. Moulding, NMFS, Feb., 2012). While toxic and biologically active substances impair corals, their effects are largely "silent," causing chronic and often sublethal stress or contributing to mortality of unapparent cause.

Physical Impacts

Coral reefs must endure physical damage from many different sources and threats acting over a range of spatial and temporal scales. Extreme wave events, such as those generated by severe tropical hurricanes, are naturally occurring processes that are typically viewed as acute disturbances. Direct physical effects from vessel groundings, anchor damage, and coastal construction activities, such as dredging, mining, and drilling, are somewhat analogous to storm damage in that they are relatively discrete events, although they generally occur over much smaller spatial scales than do storms. Other human-induced disturbances, such as those caused by tourism and recreational events, fishing gear, and marine debris, can have pervasive, chronic physical consequences. Chronic stresses reduce the ability of corals to recover from acute events (Connell et al. 1997). The relationships between injury interval and time required for reef recovery are the primary factors in evaluating equilibrium of the system (Connell 1978).

Staghorn Corals

Staghorn corals displayed severe impacts in the 1998 and 2005 bleaching events, and high temperatures and bleaching have been correlated with coral disease. The shallow reef habitat in which staghorn corals grow is especially vulnerable to increasing air and sea temperatures that accompany global climate change.

Laboratory experiments have shown that acidification reduces skeletal deposition and initiation of calcification in newly settled corals (Albright et al. 2008; Cohen et al. 2007; Cohen et al.

2009). Some CCA species provide chemical cues for settlement and enhanced post-settlement survivorship of *Acropora* larvae (Harrington et al. 2004; Ritson-Williams et al. 2010), suggesting a potential further reduction in recruitment as acidification impacts CCA growth.

White band disease is believed to be the main cause of the initial region-wide decline of staghorn corals (Aronson and Precht 2001), and disease continues to be a major threat to the 2 species. A transmissible disease termed rapid tissue loss affects staghorn coral (Williams and Miller 2005). Additionally, staghorn corals are affected by ciliates (a group of protozoans characterized by the presence of hair-like organelles; [Croquer et al. 2006]). .

Predation is a threat to staghorn corals both through direct removal of tissue and through indirect effects. Known predators include snails (*Coralliophila abbreviata*), fireworms (*Hermodice carunculata*), 2 species of damselfishes (*Stegastes planifrons* and *Microspathodon chrysurus*), and the stoplight parrotfish (*Sparisoma viride*). All of these predators are generalists, feeding on a wide range of coral species, and in some cases algae. Predation effects are more pronounced in areas where staghorn coral abundance or colony sizes are reduced, and predation pressure remains constant.

Staghorn corals appear to be particularly sensitive to sediment deposition and shading effects from increased sediment. Because they are highly dependent upon sunlight for nourishment (Lewis 1977; Porter 1976), staghorn corals are very susceptible to increases in water turbidity. Staghorn corals have poor capacity to remove coarser sediments (250-2000 μm) and only slightly more capacity for removing finer sediments (62-250 μm) (Hubbard and Pocock 1972). Water movement (turbulence) and gravity are probably more important in removing sediments from these species than their capabilities of sloughing sediments in still water (Porter 1987). A sedimentation rate of 200 mg cm^{-2} can cause both lethal (Rogers 1983) and sublethal damage resulting in compromised coral health (Hodel and Vargas-Angel 2007) in this species.

Nutrients impact staghorn corals both directly and indirectly. Nutrients from land-based sources of pollution can cause habitat loss through the stimulation of growth of algae that can occupy space on the reef (Lapointe et al. 2005). Increased levels of nutrients also reduce growth rates in staghorn corals (Renegar and Riegl 2005) and compromise their health (Hodel and Vargas-Angel 2007).

Staghorn corals are sensitive to chemical contaminants. Staghorn coral displayed higher susceptibility to copper toxicity than 2 other coral species tested; effects included depressed photosynthesis, decreased growth, tissue accumulation, and other physiological changes at exposures as low as 4 μg L^{-1} (Bielmyer et al. 2010). Staghorn coral treated with various compounds found in common sunscreens experienced rapid and complete bleaching, even at extremely low concentrations (Danovaro et al. 2008). The response of staghorn coral exposed to drilling muds produced during offshore oil and gas exploration included reduced calcification and reduced tissue soluble protein levels (Kendall et al. 1983).

The branching morphology of staghorn corals makes them particularly vulnerable to physical damage. Major storm events are a natural threat to staghorn corals that result in local population declines (Rogers et al. 1982; Woodley et al. 1981). There are observations from diverse geographical locations of coral disease outbreaks following hurricane disturbances including

Puerto Rico, (Bruckner and Bruckner 1997), Navassa, the Florida Keys, (Miller and Williams 2007; Williams et al. 2008), Bonaire, Curaçao, (*Acropora* Biological Review Team 2005), and Honduras (Halley et al. 2001). Historically, tropical storms likely fostered propagation of staghorn coral thickets through fragmentation, but recent observations from periods of frequent hurricane impacts in the Florida Keys document a lack of successful recruitment of fragments and a severe population decline (Williams et al. 2008). Staghorn corals are less able to successfully reproduce asexually due to high mortality of fragments, and reduced colony density and reef rugosity (Alvarez-Filip et al. 2009) that lessen the likelihood of retaining storm-generated fragments in suitable habitat (Williams et al. 2008). Man-made abrasion and breakage impacts to reefs are chronic and cumulative, and occur on an ongoing basis (e.g., derelict fishing gear, vessel grounding and anchoring, fishing, diver interaction).

Orbicella annularis, Orbicella faveolata, and Orbicella franksi

Because *Orbicella annularis* complex species have traditionally been common and are among the main reef builders in the Caribbean, they have been the frequent subject of research, including responses to and impacts of environmental threats. Published reports of individual bleaching surveys have consistently indicated that *O. faveolata*, *O. annularis*, and the *Orbicella annularis* complex are highly-to-moderately susceptible to bleaching (Brandt 2009; Bruckner and Hill 2009; Oxenford et al. 2008; Wagner et al. 2010). Bleaching can prevent gamete production in *O. annularis* (Mendes and Woodley 2002) and *Orbicella annularis* complex colonies (Szmant and Gassman 1990) in the following reproductive season even after they recover normal pigmentation . Bleaching events leave permanent marks in coral growth records (Leder et al. 1991; Mendes and Woodley 2002). Particularly well-documented mortalities in these species following severe mass-bleaching in 2005 highlight the immense impact that thermal stress events and their aftermath can have on *Orbicella annularis* complex populations (Miller et al. 2009). Using demographic data collected in Puerto Rico over 9 years straddling the 2005 bleaching event, Hernández-Pacheco et al. (2011) showed that population growth rates of *O. annularis* were stable in the pre-bleaching period (2001-2005), but declined in the 2 years following the bleaching event. Simulation modeling of different bleaching probabilities predicted extinction of a population with these dynamics within 100 years at a bleaching probability between 10% and 20%; in other words, once every 5-10 years (Hernández-Pacheco et al. 2011). Cervino et al. (2004) also showed that higher temperatures (over experimental treatments from 20°C-31°C) resulted in faster rates of tissue loss and higher mortality in yellow-band affected *Orbicella annularis* complex. Recent work in the Mesoamerican reef system indicated that *Orbicella faveolata* had reduced thermal tolerances in many locations and over time (Carilli et al. 2010) with increasing human populations, implying increasing local threats (Carilli et al. 2009).

The only study conducted regarding the impact of acidification on this genus is a field study that did not find any change in *Orbicella faveolata* calcification in sampled colonies from the Florida Keys up through 1996 (Helmle et al. 2011). Preliminary experiments testing effects of acidification on fertilization and settlement success of *Orbicella annularis* complex (Albright et al., unpublished data) show results that are consistent with the significant impairments demonstrated for *Acropora palmata* (Albright et al. 2010).

Both Bruckner and Hill (2009) and Miller et al. (2009) demonstrated profound declines for *Orbicella annularis* complex from disease impacts, both with and without prior bleaching. Both white-plague and yellow-band diseases can invoke this type of population level decline. Disease outbreaks can persist for years in a population; *Orbicella annularis* colonies suffering from yellow-band in Puerto Rico in 1999 still manifested similar disease signs 4 years later, with a mean tissue loss of 60% (Bruckner and Bruckner 2006).

Orbicella annularis complex does not suffer from catastrophic outbreaks of predators. While *Orbicella annularis* complex can host large populations of corallivorous snails, they rarely display large feeding scars that are apparent on other coral prey, possibly related to differences in tissue characteristics or nutritional value (Baums et al. 2003). However, low-level predation can have interactive effects with other stressors. For example, predation by butterflyfish can serve as a vector to facilitate infection of *Orbicella faveolata* with black-band disease (Aeby and Santavy 2006). Parrotfishes are also known to preferentially target *Orbicella annularis* , *O. franski,* and *O. faveolata* in so-called "spot-biting," which can leave dramatic signs in some local areas (Bruckner et al. 2000; Rotjan and Lewis 2006). Chronic parrotfish biting can impede colony recovery from bleaching in *O. franksi* and *O. faveolata* (Rotjan et al. 2006). Although it is not predation per se, *Orbicella* colonies have often been infested by other pest organisms. Bio-eroding sponges (Ward and Risk 1977) and territorial damselfishes, *Stegastes planifrons*, can cause tissue loss and skeletal damage. Damselfish infestation of *Orbicella annularis* complex appears to have increased in areas where their preferred, branching coral habitat has declined because of loss of Caribbean acroporids (Precht et al. 2010).

Large, massive, long-lived colonies of *Orbicella annularis* complex lend themselves to retrospective studies of coral growth in different environments, so there is a relatively large amount known or inferred regarding relationships between water quality and *Orbicella annularis* complex growth and status. For example, Tomascik (1990) found an increasing average growth (linear extension) rate of *Orbicella annularis* complex with improving environmental conditions on fringing reefs in Barbados. Within the same study, Tomascik also found a general pattern of decreasing growth rates within the past 30 years at each of the 7 fringing reefs and contributed this decrease to the deterioration of water quality along the west coast of Barbados. Torres and Morelock (2002) noted a similar decline in *Orbicella annularis* complex growth at sediment-impacted reefs in Puerto Rico. Density and calcification rate increased from high to low turbidity and sediment load, while extension rate followed an inverse trend (Carricart-Ganivet and Merino 2001). Eakin et al. (1994) demonstrated declines in *Orbicella annularis* linear extension during periods of construction in Aruba. Downs et al. (2005) suggested that localized toxicant exposure may account for a localized mortality event of *Orbicella annularis* complex in Biscayne National Park. *Orbicella faveolata* had somewhat lesser sensitivity to copper exposure in laboratory assays than *Acropora cervicornis* and *Pocillopora damicornis* (Bielmyer et al. 2010). Nutrient-related runoff has also been deleterious to *Orbicella annularis* complex. Elevated nitrogen reduced respiration and calcification in *Orbicella annularis* and stimulated zooxanthellae populations (Marubini and Davies 1996). Elevated nutrients increased the rate of tissue loss in *Orbicella franksi* and *Orbicella faveolata* affected by yellow-band disease (Bruno et al. 2003). Chronic nutrient elevation can produce bleaching and partial mortality in *Orbicella annularis*, whereas anthropogenic dissolved organic carbon kills corals directly (Kuntz et al. 2005).

Rough Cactus Coral

Rough cactus coral is susceptible to acute and subacute white plague. Dustan (1977) reported dramatic impacts from this disease to the population in the upper Florida Keys in the mid-1970s. He also reported that the rate of disease progression was positively correlated with water temperature and measured rates of disease progression up to 3 mm per day.

The susceptibility of rough cactus corals to nutrients is unknown. However, the absence of this species at fringing reef sites impacted by sewage pollution (Tomascik and Sander 1987) suggests it is highly susceptible to nutrient over-enrichment.

No specific research has addressed the effects of acidification on the genus *Mycetophyllia*

Lamarck's Sheet Coral

Lamarck's sheet coral is susceptible to bleaching at elevated temperatures (Ghiold and Smith 1990), via direct loss of zooxanthellae as well as decreased pigment content (Porter et al. 1989). In laboratory studies in Jamaica, Lamarck's sheet coral tolerated temperatures up to 32°C (Fitt and Warner 1995) but virtually complete disruption of photosynthesis occurred at 32°C–34°C (Warner et al. 1996). Cold stress has also produced bleaching (Bak et al. 2005). Although bleaching can often be extensive, it may not induce mortality in Lamarck's sheet coral (Aronson and Precht 2000; Aronson et al. 1998; Porter et al. 1989).

No specific research has addressed the effects of acidification on the genus *Agaricia*.

Lamarck's sheet coral is vulnerable to white plague disease (Garzon-Ferreira et al. 2001; Nugues 2002; Richardson 1998), ciliate infections (Croquer et al. 2006), and tumors (UNEP 2010). The ecological and population impacts of disease have not been established for *Agaricia lamarcki*.

The effects of land-based sources of pollution (LBSP) on the genus *Agaricia* are largely unknown. *Agaricia* sp. typically have small calices (i.e., skeletal structure in which the coral polyp sits) and are not efficient sediment rejecters (Hubbard and Pocock 1972). *Agaricia lamarcki*'s platy morphology could make it sediment-susceptible. Vertical plates of *Agaricia* shed more sediment than horizontally-oriented ones (Bak and Elgershuizen 1976), and fine sediment suspended in hurricanes can cause much higher mortality in platy corals than hemispherical or non-flat morphologies (Bak, unpublished data; Bak et al. 2005).

Elliptical Star Coral

Although elliptical star coral is susceptible to bleaching, it showed the lowest bleaching response of species observed to bleach in the south Florida region (Wagner et al. 2010). In Barbados elliptical star coral ranked 16th of 21 species in bleaching prevalence during the 2005 Caribbean mass-bleaching event (Oxenford et al. 2008). It was also observed to be bleaching-tolerant in the U.S. Virgin Islands during the same event (Clark et al. 2009). Hence, this species is regarded to be at relatively low threat from temperature-induced bleaching. Elliptical star coral hosts clade B zooxanthellae (Correa et al. 2009; LaJeunesse 2002). Zooxanthellae in clade B do not grow well at high temperatures (Kinzie et al. 2001), but in the field, corals with this clade may be relatively bleaching-resistant (McField 1999). Experimental studies suggest clade B is more bleaching-resistant than clade C, but less resistant than clade A (Warner et al. 2006).

No specific research has addressed the effects of acidification on the genus *Dichocoenia*.

Elliptical star coral is highly susceptible to white plague, with infection increasing with temperature (Borger and Steiner 2005). An outbreak event for this disease in the Florida Keys had demonstrable impact at the local population level, yielding mortality of 75% of colonies across several reef sites, substantial shifts in population structure, and essentially no recovery over a 7-year follow-up period (Richardson and Voss 2005). This species is also susceptible to black-band disease (Sutherland et al. 2004), ciliate infection (Croquer et al. 2006), and dark-spot syndrome (Borger and Steiner 2005). Disease susceptibility appears to be variable (Borger and Steiner 2005); for example, *Dichocoenia stokesi* was minimally affected during a 1998 outbreak in St. Lucia that caused widespread mortality in *Orbicella faveolata* and other species (Nugues 2002).

Elliptical star coral is minimally affected by predation. It can be heavily bioeroded, particularly by bivalves (Highsmith 1981), and lose substantial amounts of tissue to sponge overgrowth (Hill 1998).

One laboratory study (Telesnicki and Goldberg 1995b) has shown that elliptical star coral displays physiological stress at turbidity levels that are within allowable levels as regulated by the State of Florida for coastal construction projects. While light levels and photosynthesis were not affected, respiration levels and mucous production were significantly higher at turbidity levels as low as 14–16 Nephelometric Turbidity Units (NTU), and photosynthesis to respiration ratio fell below 1 at 28–30 NTU (Telesnicki and Goldberg 1995a). An earlier laboratory study examining oil/sediment rejection indicated that elliptical star coral was intermediate (of 19 Caribbean coral species examined) in the rate of sediment removal from its tissues (Bak and Elgershuizen 1976).

4.2.3 Elkhorn and Staghorn Coral Designated Critical Habitat

Elkhorn and staghorn corals require hard, consolidated substrate, including attached, dead coral skeleton, for their larvae to settle. Within the geographical area occupied by a listed species, critical habitat consists of specific areas on which those physical or biological features essential to the conservation of the species are found. For elkhorn and staghorn coral, the physical feature of critical habitat essential to the conservation of the species is substrate of suitable quality and availability, in water depths from the mean high water line to 30 m, to support successful larval settlement, recruitment, and reattachment of fragments. Substrate of suitable quality and availability means consolidated hardbottom or dead coral skeletons free from fleshy and turf macroalgae, and sediment cover. A shift in benthic community structure from coral-dominated to algae-dominated that has been documented since the 1980s means that the settlement of larvae or attachment of fragments is often unsuccessful (Hughes and Connell 1999). Sediment accumulation on suitable substrate also impedes sexual and asexual reproductive success by preempting available substrate and smothering coral recruits.

While algae, including crustose coralline algae and fleshy macroalgae, are natural components of healthy reef ecosystems, increases in the dominance of algae since the 1980s impedes coral recruitment. The overexploitation of grazers through fishing has also enabled fleshy macroalgae to persist in reef and hardbottom areas formerly dominated by corals. Impacts to water quality, in particular nutrient inputs, associated with coastal development are also thought to enhance the growth of fleshy macroalgae by providing them with nutrient sources. Fleshy macroalgae are able to colonize dead coral skeleton and other hard substrate and some are able to overgrow living corals and crustose coralline algae. Because crustose coralline algae is thought to provide chemical cues to coral larvae indicating an area is appropriate for settlement, overgrowth by macroalgae may affect coral recruitment (Steneck 1986). Several studies show that coral recruitment tends to be greater when algal biomass is low (Rogers et al. 1984, Hughes 1985, Connell et al. 1997, Edmunds et al. 2004, Birrell et al. 2005, Vermeij 2006). In addition to preempting space for coral larval settlement, many fleshy macroalgae produce secondary metabolites with generalized toxicity, which also may inhibit settlement of coral larvae (Kuffner and Paul 2004). The rate of sediment input from natural and anthropogenic sources can affect reef distribution, structure, growth, and recruitment. Sediments can accumulate on dead and living corals and exposed hardbottom, thus reducing the available substrate for larval settlement and fragment attachment.

In addition to the amount of sedimentation, the source of sediments can affect coral growth. In a study of 3 sites in Puerto Rico, Torres (2001) found that low-density coral skeleton growth was correlated with increased resuspended sediment rates and greater percentage composition of terrigenous sediment. In sites with higher carbonate percentages and corresponding low percentages of terrigenous sediments, growth rates were higher. This suggests that re-suspension of sediments and sediment production within the reef environment does not necessarily have a negative impact on coral growth while sediments from terrestrial sources increase the probability that coral growth will decrease, possibly because terrigenous sediments do not contain minerals that corals need to grow (Torres 2001).

Long-term monitoring of sites in the U.S.V.I. indicate that coral cover has declined dramatically; coral diseases have become more numerous and prevalent; macroalgal cover has increased; fish of some species are smaller, less numerous, or rare; long-spined black sea urchins are not abundant; and sedimentation rates in nearshore waters have increased from one to 2 orders of magnitude over the past 15 to 25 years (Rogers et al. 2008). Thus, changes that have affected elkhorn and staghorn coral and led to significant decreases in the numbers and cover of these species have also affected the suitability and availability of habitat.

Figure 10**,** below, shows the boundaries of the Florida area of *Acropora* critical habitat. The Florida area contains 3 sub-areas. The shoreward boundary for Florida sub-area A begins at the 6-ft (1.8 m) contour at the south side of Boynton Inlet, Palm Beach County at 26° 32′ 42.5″ N; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with latitude 25° 45′ 55″ N, Government Cut, Miami-Dade County; then runs due west to the point of intersection with the 6-ft (1.8 m) contour, then follows the 6-ft (1.8 m) contour to the beginning point. The shoreward boundary of Florida sub-area B begins at the MLW line at 25° 45′ 55″ N, Government Cut, Miami-Dade County; then runs due east to the point of intersection with the 98-ft (30 m) contour; then follows the 98-ft (30 m) contour to the point of intersection with longitude 82°W; then runs due north to the point of intersection with the South Atlantic Fishery Management Council (SAFMC) boundary at 24° 31′ 35.75″ N; then follows the SAFMC boundary to a point of intersection with the MLW line at Key West, Monroe County; then follows the MLW line, the SAFMC boundary (see 50 CFR 600.105(c)), and the COLREGS line (see 33 CFR 80.727. 730, 735, and 740) to the beginning point. The seaward boundary of Florida sub-area C (the Dry Tortugas) begins at the northern intersection of the 98-ft (30 m) contour and longitude 82° 45’ W; then follows the 98–ft (30 m) contour west around the Dry Tortugas, to the southern point of intersection with longitude 82° 45’ W; then runs due north to the beginning point.

Critical habitat does not include the following particular areas: (1) all areas subject to the 2008 Naval Air Station Key West Integrated Natural Resources Management Plan, (2) all areas containing existing (already constructed) federally authorized or permitted man-made structures such as aids-to-navigation (ATONs), artificial reefs, boat ramps, docks, pilings, maintained channels, or marinas, (3) all waters identified as existing (already constructed) federally authorized channels, and (4) all waters of the Restricted Anchorage Area as described at 33 CFR 334.580, beginning at a point located at 26° 05′ 30’’ N, 80 03′ 30’’ W.; proceed west to 26° 05′ 30″ N, 80° 06′ 30″ W; thence, southerly to 26° 03′ 00″ N, longitude 80° 06′ 42″ W; thence, east to latitude 26° 03′ 00″ N, 80° 05′ 44″ W.; thence, south to 26° 01′ 36″ N, 80° 05′ 44″ W.; thence, east to 26° 01′ 36″ N, 80° 03′ 30″ W; thence, north to the point of beginning.

The proposed project takes place in sub-area B within the Florida area of critical habitat. The entire Florida area is comprised of 1,329 square miles of designated critical habitat.

Threats

The final critical habitat rule for elkhorn and staghorn coral identifies several sources of threat to the essential feature. Suitable habitat available for larval settlement and recruitment, and asexual fragment reattachment and recruitment of these coral species is particularly susceptible to impacts from human activity because of the shallow water depth range (less than 98 ft/30 m) in which elkhorn and staghorn corals commonly grow and the essential feature occurs. The proximity of this habitat to coastal areas subject this feature to impacts from multiple activities, including, but not limited to dredging and disposal activities, stormwater run-off, coastal and maritime construction, land development, wastewater and sewage outflow discharges, point and non-point source pollutant discharges, fishing, placement of large vessel anchorages, and installation of submerged pipelines or cables. The impacts from these activities, combined with those from natural factors (e.g., major storm events), significantly affect the quality and quantity of available substrate for these threatened species to successfully sexually and asexually reproduce.

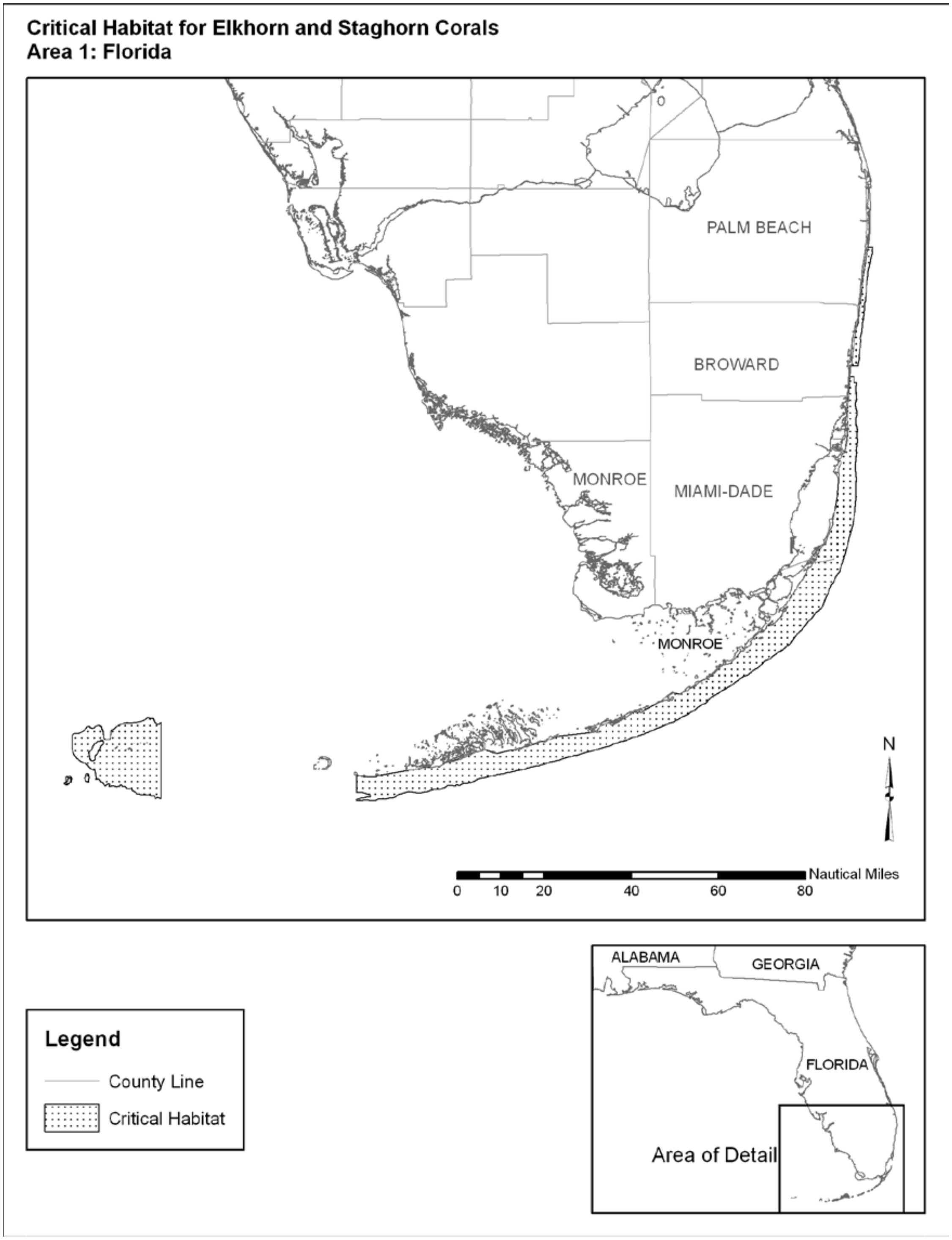

Figure 10. Florida unit designated critical habitat for *Acropora cervicornis* and *Acropora palmata* (50 CFR Parts 223 and 226 Endangered and Threatened Species; Critical Habitat for Threatened Elkhorn and Staghorn Corals; Final Rule)

4.2.4 Johnson's Seagrass

Johnson's seagrass is the first marine plant ever listed under the ESA. It was listed as "threatened" on September 14, 1998, based on the results of fieldwork and a status review initiated in 1990. Kenworthy (1993, 1997, 1999) and NMFS (2007) discuss the field study results and summarize an extensive literature review on the status of Johnson's seagrass. In addition to the published literature, the Johnson's Seagrass Recovery Implementation Team (Recovery Team) is currently updating the 2002 Recovery Plan for Johnson's Seagrass. The updated Recovery Plan is in review, but much of the information contained in this opinion that updates our knowledge of the status of and threats to the species, life history information, and cumulative impacts, comes from talks with Dr. W. Judson Kenworthy (Team Leader) and other NMFS members of the Recovery Team, and from their review of sections of the updated Recovery Plan. That information is attributed throughout this opinion to the Recovery Team. The following discussion summarizes those findings relevant to our evaluation of the proposed action.

Life History and Population Biology

Based on the current knowledge of the species, Johnson's seagrass reproduction is believed to be entirely asexual, and dispersal is by vegetative fragmentation. Sexual reproduction in Johnson's seagrass has not been documented. Female flowers have been found, although dedicated surveys in the Indian River Lagoon have not discovered male flowers, fertilized ovaries, fruits, or seeds either in the field or under laboratory conditions (Jewett-Smith et al. 1997, Hammerstrom and Kenworthy 2002, NMFS 2007). Searches throughout the range of Johnson's seagrass have produced the same results, suggesting either that the species does not reproduce sexually or that the male flowers are difficult to observe or describe, as noted for other *Halophila* species (Kenworthy 1997). Surveys to date indicate that the incidence of female flowers appears to be much higher near the inlets leading to the Atlantic Ocean.

Throughout its range, Johnson's seagrass occurs in dynamic and disjunctive patches. It spreads rapidly, growing horizontally from dense apical meristems with leaf pairs having short life spans (Kenworthy 1997). Kenworthy suggested that the observed horizontal spreading, rapid growth patterns, and high biomass turnover could explain the dynamic patches observed in distribution studies of this species. While patches may colonize quickly, they may also disappear rapidly. Sometimes they will disappear for several years and then re-establish, a process referred to as "pulsating patches" (Heidelbaugh et al. 2000, Virnstein and Morris 2007, Virnstein et al. 2009). Mortality, or the disappearance of patches, can be caused by a number of processes, including burial from bioturbation and sediment deposition (Heidelbaugh et al. 2000), erosion, herbivory, desiccation, and turbidity. In the absence of sexual reproduction, one possible explanation for the pulsating patches is dispersal and reestablishment of vegetative fragments, a process that commonly occurs in aquatic plants and has been demonstrated in other seagrasses (Philbrick and Les 1996, DiCarlo et al. 2005), and was also recently confirmed by experimental mesocosm studies with Johnson's seagrass (Hall et al. 2006).

Johnson's seagrass is a shallow-rooted species and vulnerable to uprooting by wind, waves, storm events, tidal currents, bioturbation, and motor vessels. It is also vulnerable to burial by sand movement and siltation (Heidelbaugh et al. 2000). Having a canopy of only 2-5 cm, it may be easily covered by sediments transported during storms or redistributed by macrofaunal bioturbation during the feeding activities of benthic organisms. Mesocosm experiments indicate that clonal fragments can only survive burial for up to a period of 12 days (W.J. Kenworthy, CCFHR, NOAA, Beaufort, North Carolina, unpublished). Mechanisms capable of disturbing patches may create clonal fragments that become dispersed. Hall et al. (2006) showed that drifting fragments of Johnson's seagrass can remain viable for 4 to 8 days, during which time they can settle, root, and grow. The process of asexual fragmentation can occur year-round. Fragments could drift several kilometers under the influence of wind and tidally- driven circulation, providing potential recruits for dispersal and new patch formation. In the absence of sexual reproduction, these are likely to be the most common forms of dispersal and patch maintenance.

Population Status and Distribution
Johnson's seagrass occurs in a variety of habitat types, including on intertidal wave-washed sandy shoals, on flood deltas near inlets, in deep water, in soft mud, and near the mouths of canals and rivers, where presumably water quality is sometimes poor and where salinity fluctuates widely. It is an opportunistic plant that occurs in a patchy, disjunctive distribution from the intertidal zone to depths of approximately 2-3 meters in a wide range of sediment types, salinities, and in variable water quality conditions (NMFS 2007).

Johnson's seagrass exhibits a narrow geographical range of distribution and has only been found growing along approximately 200 kilometers (km) of coastline in southeastern Florida north of Sebastian Inlet, Indian River County, south to Virginia Key in northern Biscayne Bay, Miami-Dade County. This apparent endemism suggests that Johnson's seagrass has the most limited geographic distribution of any seagrass in the world. Kenworthy (1997, 1999) confirmed its limited geographic distribution in patchy and vertically disjunctive areas throughout its range. Since the last status review (NMFS 2007), there have not been any reported reductions in the geographic range of the species. In fact, the St. Johns River Water Management District (SJRWMD) observed Johnson's seagrass approximately 21 km north of the Sebastian Inlet mouth on the western shore of the Indian River Lagoon – a discovery that slightly extends the species' known northern range (Virnstein and Hall 2009).

Two survey programs regularly monitor the presence and abundance of Johnson's seagrass within this range. One program, conducted by the SJRWMD since 1994, covers the northern section of the species' geographic range between Sebastian Inlet and Jupiter Inlet (Virnstein and Morris 2007, Virnstein et al. 2009). The second recently initiated survey (2006) is of the southern range of the species between Jupiter Inlet and Virginia Key in Biscayne Bay (Kunzelman 2007). Johnson's seagrass is a perennial species (meaning it lasts for greater than 2 growing seasons), showing no consistent seasonal or year-to-year pattern based on the northern transect surveys, but has exhibited some winter decline (NMFS 2007). However, during exceptionally mild winters, Johnson's seagrass can maintain or even increase in abundance from summer to winter. In the surveys conducted between 1994 and 2007, it occurred in 7.1% of the 1-m^2 quadrats in the northern range. Depth of occurrence within these surveys ranged from 0.03 to 2.5 m. Where it does occur, its distribution is patchy, both spatially and temporally. It frequently disappeared from transects only to reappear several months or several years later (NMFS 2007).

Based on the results of the southern transect sampling, it appears there is a relatively continuous, although patchy, distribution of the species from Jupiter Inlet to Virginia Key (NMFS 2007). The largest reported contiguous patch of Johnson's seagrass in the southern range was observed in Lake Worth Lagoon and was estimated to be 30 acres (Kenworthy 1997). Eiseman and McMillan (1980) documented Johnson's seagrass in the vicinity of Virginia Key (latitude 25.75° N); this location is considered to be the southern limit of the species' range. There have been no reports of this species farther south of the currently known southern distribution. The presence of Johnson's seagrass in northern Biscayne Bay (north of Virginia Key) is well documented. In addition to localized surveys, the presence of Johnson's seagrass has been documented by various field experiences and observations of the area by federal, state, and county entities. Johnson's seagrass has been documented in various USACE and USCG permit applications reviewed by NMFS. Findings from the southern transect sampling (summer 2006 and winter 2007) show little difference in the species' frequency or abundance between the summer and winter sampling period. The lower frequencies of Johnson's seagrass occurred at those sites where larger-bodied seagrasses (e.g., *Thalassia testudinum* [turtle grass] and *Syringodium filiforme* [manatee grass]) were more abundant (NMFS 2007). The southern range transect data support some of the conclusions drawn from previous studies and other surveys. This is a rare species; however, it can be found in relatively high abundance where it does occur. Based on the results of the southern transect sampling, it appears that, although it is disjunctively distributed and patchy, there is some continuity in the southern distribution, at least during periods of relatively good environmental conditions and no significant large-scale disturbances (NMFS 2007).

Information on the species' distribution and results of limited experimental work suggest that Johnson's seagrass has a wider tolerance range for salinity, temperature, and optical water quality conditions than other species such as paddle grass, *Halophila decipiens* (Dawes et al. 1989, Kenworthy and Haunert 1991, Gallegos and Kenworthy 1996, Kenworthy and Fonseca 1996, Durako et al. 2003, Kunzelman et al. 2005, Torquemada et al. 2005). Johnson's seagrass has been observed near the mouths of freshwater discharge canals (Gallegos and Kenworthy 1996), in deeper turbid waters of the interior portion of the Indian River Lagoon (Kenworthy 2000, Virnstein and Morris 2007), and in clear water associated with the high energy environments and flood deltas inside ocean inlets (Kenworthy 1993, 1997; Virnstein et al. 1997; Heidelbaugh et al. 2000; Virnstein and Morris 2007). It can colonize and persist in high- tidal-energy environments and has been observed where tidal velocities approach the threshold of motion for unconsolidated sediments (35-40 cm s^{-1}). The persistent presence of high-density, elevated patches of Johnson's seagrass on flood tidal deltas near inlets suggests that it is capable of sediment stabilization. Intertidal populations of Johnson's seagrass may be completely exposed at low tides, suggesting high tolerance to desiccation and wide temperature tolerance.

In Virnstein's study areas within the Indian River Lagoon, Johnson's seagrass was found associated with other seagrass species or growing alone in the intertidal, and, more commonly, at the deep edge of some transects in water depths down to 180 cm. In areas in which long-term poor water and sediment quality have existed until recently, Johnson's seagrass appears to occur in relatively higher abundance, perhaps due to the inability of the larger species to thrive. Johnson's seagrass appears to be out-competed in seagrass habitats where environmental conditions permit the larger seagrass species to thrive (Virnstein et al. 1997, Kenworthy 1997). When the larger, canopy-forming species are absent, Johnson's seagrass can grow throughout the full seagrass depth range of the Indian River Lagoon (NMFS 2007, Virnstein et al. 2009).

Observations by researchers have suggested that Johnson's seagrass exploits unstable environments or newly-created un-vegetated patches by exhibiting fast-growth and support for all local ramets in order to exploit areas in which it could not otherwise compete. It may quickly recruit to locally uninhabited patches through prolific lateral branching and fast horizontal growth. While these attributes may allow it to compete effectively in periodically disturbed areas, if the distribution of this species becomes limited to stable areas it may eventually be out-competed by more stable-selected plants represented by the larger-bodied seagrasses (Durako et al. 2003). In addition, the physiological attributes of Johnson's seagrass may limit growth (i.e., spreading) over large areas of substrate if the substrate is somehow altered (e.g., dredged to a depth that would preclude future recruitment of Johnson's seagrass); therefore, its ability to recover from widespread habitat loss may be limited. The clonal and reproductive growth characteristics of Johnson's seagrass result in its distribution being patchy, noncontiguous, and temporally fluctuating. These attributes suggest that colonization between broadly disjunctive areas is likely difficult and that the species' risk of extinction may be increased if it is removed from large areas within its range by natural or anthropogenic means.

Threats

The emerging consensus among seagrass experts on the Recovery Team is that the possibility of mortality due to reduced salinity over long periods of time is the most clearly identified threat to the species' long-term persistence. Some studies have shown that Johnson's seagrass has a wide tolerance for salinity. However, short-term experiments have shown reduced photosynthesis and increased mortality at low salinities (<10 psu [practical salinity units=parts per thousand]). Longer duration mesocosm experiments have resulted in 100% mortality of Johnson's seagrass after 10 days at salinities <10 psu (Kahn and Durako 2008). The Recovery Team has recently determined that the most significant threat to the species is the present or threatened destruction, modification or curtailment of its habitat or range through water management practices and stochastic environmental factors which can alter the salinity of its habitat. Given that it is not uncommon for salinities to decline below 15-20 psu in its range (Steward et al. 2006), and that a number of natural and human-related factors can affect salinity throughout its range, the Recovery Team identified reduced salinity as a potentially significant threat to the species because the potential for long-term mortality over a large scale could counteract the life history strategy the species uses to persist in the face of numerous, ongoing environmental impacts. In previous reviews, including the critical habitat listing rule and the 2002 Recovery Plan, several additional factors were considered threats, including: (1) dredging and filling, (2) construction and shading from in- and over-water structures, (3) propeller scarring and anchor mooring, (4) trampling, (5) storms, and (6) siltation. In reviewing all information available since the original listing, the Recovery Team conducted assessments of each of these factors and has been unable to confirm that any of these poses a significant threat to the persistence and recovery of the species. A brief discussion of these factors follows.

Routine maintenance dredging associated with the constant movement of sediments in and around inlets may affect seagrasses by direct removal, light limitation due to turbidity, and burial from sedimentation. The disturbance of sediments can also destabilize the benthic community. Altering benthic topography or burying the plants may remove them from the photic zone. Permitted dredging of channels, basins, and other in- and on-water construction projects cause loss of Johnson's seagrass and its habitat through direct removal of the plants, fragmentation of habitat, shading, turbidity, and sedimentation. Although dredge and fill activities can and do adversely affect Johnson's seagrass and its designated critical habitat, federal, state, and local permitting programs closely scrutinize these activities and the construction of in- and over-water structures. The USACE, under Section 404 of the Clean Water Act and Section 10 of the Rivers and Harbors Act, has federal authority over the issuance of dredge-and-fill permits. This permitting process includes language to protect and conserve seagrasses through field evaluations, consultations, and recommendations to avoid, minimize, and mitigate for impacts to seagrasses.

Height, width, and orientation have been identified as the 3 most important factors affecting seagrass growth and abundance under and around over-water structures (Burdick and Short 1999, Beal and Schmit 2000). Landry et al. (2008) stated there is a compelling argument supporting prior studies that indicate that docks can have negative impacts on seagrasses by reducing their abundance and in some cases, preventing seagrass from growing. Their study found evidence that all species of seagrass were impacted by docks. However, they found that although it is reduced in frequency under grated docks, Johnson's seagrass was observed in higher densities under the grated docks compared to non-grated docks. Furthermore, their results suggest that Johnson's seagrass does benefit from the light-transmitting characteristics of grated decking. Landry et al. (2008) found that grated docks were more similar to the adjacent and the reference transects (for seagrass) than non-grated docks. This suggests that while both grated and non-grated docks can have detrimental effects on seagrass beds, grated docks are relatively less detrimental to seagrass beds than non-grated docks. Given the supporting experimental evidence that fiberglass grating does improve the incident solar radiation penetrating under structures (Shafer and Robinson 2001), continuing to require grated decking will benefit most seagrasses. Landry et al. (2008) recommend that grated decking should be used for any dock construction to take place over seagrasses, most importantly Johnson's seagrass.

In the results from their study evaluating the regulatory construction guidelines to minimize impacts to seagrasses from single-family residential dock structures in Florida and Puerto Rico, Shafer et al. (2008) emphasized avoidance of seagrasses as a first priority. Avoidance may be achieved by relocating or realigning the structure. It is important to note that Shafer et al. (2008) observed that in the majority of cases, permit applicants and regulatory agencies are, when practicable, generally succeeding in avoiding seagrass impacts by extending the length of the access walkway so that the terminal platform is constructed in deep water that is not conducive to seagrass growth. If avoidance is not possible, Shafer et al. (2008) recommend revising the USACE-NMFS dock construction guidelines to prioritize dock orientation (in a north-south direction) and height (minimum of 5 ft above mean high water) as the most important specifications for the survivorship of seagrasses under docks.

Most dock construction is subject to the construction guidelines (i.e., the USACE and NMFS jointly developed *Key for Construction Conditions for Docks or Other Minor Structures Constructed in or over Johnson's Seagrass* ("Johnsons seagrass key"), dated October 2002 and the associated publication, *Dock Construction Guidelines in Florida for Docks or Other Minor Structures Constructed in or over Submerged Aquatic Vegetation, Marsh, or Mangrove Habitat* ("dock construction guidelines"), dated August 2001. Some docks meeting certain provisions, are exempt from state permitting (http://www.dep.state.fl.us/central/Home/SLERP/Docks/sfdock.pdf) and contribute to loss of Johnson's seagrass through construction impacts and shading.

The USACE's State (Florida) Programmatic General Permit Program (SPGP IV-R1) authorizes permits for in-water construction activities that include: shoreline stabilization projects; construction of boat ramps, boat launch areas and structures associated with such ramps or launch areas; docks, pier associated facilities, and other minor piling-supported structures, and; maintenance dredging of canals and channels. An increasing number of docks in Florida are now permitted through the SPGP. From January 1, 2000-March 31, 2009, the SPGP was utilized 19,927 times of which 52% of this total was for single-family docks. The SPGP does not allow construction in Johnson's seagrass critical habitat. For a dock to be authorized under the SPGP, the applicant must fully comply with the *Johnson's Seagrass Key* and the associated *Dock Construction Guidelines*. Additional project design criteria apply to the SPGP (e.g., docks must be ≤ 1000 sq ft). Similarly, the USACE's SAJ-42 permit allows Miami-Dade County to authorize permits for minor dredging and construction projects within the county. The projects authorized under the SAJ-42 permit must comply with *"Johnson's Seagrass Key"* and the associated *"Dock Construction Guidelines*." No docks were authorized from April 2006 to April 2011 inside of Johnson's seagrass critical habitat outside of Miami-Dade County and only 8 dock/pier projects were authorized in counties within JSG range, but outside of critical habitat (Broward County = 3, Palm Beach County = 3, and 2 in Martin County). Lastly, NMFS recently completed a programmatic consultation on 12 SAJ general permits. The 12 SAJ consultation is for the entire state of Florida, and covered permits issued by the USACE authorize small maintenance dredging, private single-family boat ramps, aerial transmission lines, other minor structures, single-family docks, private multi-family docks, commercial docks, and bulkheads and backfills as long as they meet certain size requirements and limitations on construction methodology.

The Recovery Team has identified weaknesses in the oversight practices of state and federal agencies in the permitting process for some or all of the activities discussed above, due to budget, staffing, and technological limitations. The need for post-construction permit compliance and enforcement for dock structures in Florida and Puerto Rico has been discussed in Shafer et al. (2008). The Recovery Team also identified difficulties in monitoring a rare and patchily-distributed species in single-event surveys associated with permit applications and continues to work with collaborators to improve monitoring methods. The Recovery Team has worked with NMFS's Protected Resources and Habitat Conservation staff to develop and improve guidelines for site monitoring methods (Greening and Holland 2003), dock construction guidelines (NMFS and USACE 2002, Shafer et al. 2008), and best management practices to minimize the impact of docks on Johnson's seagrass (Landry et al. 2008). While it is recognized that dredging and filling and construction and shading from in- and over-water structures can adversely affect Johnson's seagrass and its habitat, the Recovery Team determined that these activities are typically local and small-scale and the deficiencies in the permitting process were not presently a significant threat to the survival of Johnson's seagrass because they will not individually or cumulatively result in long-term, large-scale mortality of Johnson's seagrass, and preclude the species from its strategy of recolonizing areas.

Propeller scarring and improper anchoring are known to adversely affect seagrasses (Sargent et al. 1995, Kenworthy et al. 2002). These activities can severely disrupt the benthic habitat by uprooting plants, severing rhizomes, destabilizing sediments, and significantly reducing the viability of the seagrass community. Propeller dredging and improper anchoring in shallow areas are a major disturbance to even the most robust seagrasses. This destruction is expected to worsen with the predicted increase in boating activity within Florida. The Florida Department of Motor Vehicles reported a total of 1,027,043 registered commercial and recreational vessels statewide in 2007, a peak after years of growth. Registrations declined slightly subsequently, likely due to the economic downturn, to 982,470 in 2009 (DHSMV 2010). This number is likely to increase based on Florida's projected population growth of 18 million in 2006 to 25 million in 2025 (http://www.propertytaxform.state.fl/docs/eo06141.pdf). An increase in the number of registered vessels will likely lead to an increase in adverse effects to seagrasses caused by propeller dredging/scarring. Other indirect effects associated with motor vessels include turbidity from operating in shallow water, dock construction and maintenance, marina expansion, and inlet maintenance dredging. These activities and impacts are also likely to increase (NMFS 2007). Damage to seagrasses from propeller scarring and improper anchoring by motor vessels is recognized as a significant resource management problem in Florida (Sargent et al. 1995). A number of local, state, and federal statutes protect seagrasses from damage due to vessel impacts, and a number of conservation measures, including the designation of vessel control zones, signage, mooring fields, and public awareness campaigns, are directed at minimizing vessel damage to seagrasses. Despite these efforts, vessel damage can have significant local and small-scale (1 m^2 to 100 m^2) impacts on seagrasses (Kirsch et al. 2005), but there is no direct evidence that these small-scale local effects are so widespread that they are a threat to the persistence and recovery of Johnson's seagrass.

Trampling of seagrass beds, a secondary effect of recreational boating, also disturbs seagrass habitat, but is a lesser concern. Trampling damages seagrasses by pushing leaves into the sediment and crushing or breaking the leaves and rhizomes. Since the designation of critical habitat, however, there have been no documented observations or reports of damage by trampling, and if there was, it would be small-scale and local. Therefore, the Recovery Team determined that trampling does not constitute a significant threat to the survival or recovery of Johnson's seagrass.

Large-scale weather events, such as tropical storms and hurricanes, while they often generate runoff conditions that decrease water quality, they also produce conditions (wind setup and abrupt water elevation changes) that can increase flushing rates. The effects of storms can be complex. Specifically documented storm effects on seagrasses include: (1) scouring and erosion of sediments, (2) erosion of seeds and plants by waves, currents, and surge, (3) burial by shifting sand, (4) turbidity, and (5) discharge of freshwater, including inorganic and organic constituents in the effluents (Steward et al. 2006). Storm effects may be chronic, e.g., due to seasonal weather cycles, or acute, such as the effects of strong thunderstorms or tropical cyclones. Studies have demonstrated that healthy, intact seagrass meadows are generally resistant to physical degradation from severe storms, whereas damaged seagrass beds may not be as resilient (Fonseca et al. 2000, Whitfield et al. 2002). In the late summer and early fall of 2004, 4 hurricanes passed directly over the northern range of Johnson's seagrass in the Indian River Lagoon. A post-hurricane random survey in the area of the Indian River Lagoon affected by the 4 hurricanes indicated the presence of Johnson's seagrass was similar to that reported by the SJRWMD transect surveys prior to the storms. This indicates that while the species may temporarily decline, under the right conditions it can return quickly (Virnstein and Morris 2007). Furthermore, despite evidence of longer-term reductions in salinity, increased water turbidity, and increased water color associated with higher than average precipitation in the spring of 2005, there was no evidence of long-term chronic impacts to seagrasses and no direct evidence of damage to Johnson's seagrass that could be considered a threat to the survival of the species (Steward et al. 2006).

Silt derived from adjacent land and shoreline erosion, river and canal discharges, inlets, and internally re-suspended materials can lead to the accumulation of material on plant leaves causing light deprivation. Deposition of silt can also lead to the burial of plants, accumulation of organic matter, and anoxic sediments. Johnson's seagrass grows in a wide range of environments, including those that are exposed to siltation from all the potential sources. Documentation of the direct effects of siltation on seagrasses are generally unavailable. The absence of seagrass has been associated with the formation of muck deposits, however, and localized areas of flocculent, anoxic sediments in isolated basins and segments of the Indian River Lagoon have been observed. Furthermore, sustained siltation experimentally simulated by complete burial for at least 12 days may cause mortality of Johnson's seagrass (W.J. Kenworthy, CCFHR, NOS, Beaufort, North Carolina, unpublished data). In general, the effects of siltation are localized and not widespread and are not likely to threaten the survival of the species.

In addition to the 6 factors discussed above, we also consider the effects of altered water quality on Johnson's seagrass. Availability of light is one of the most significant environmental factors affecting the survival, growth, and distribution of seagrasses (Bulthuis 1983, Dennison 1987, Abal et al. 1994, Kenworthy and Fonseca 1996). Water quality and the penetration of light are affected by turbidity (suspended solids), color, nutrients, and chlorophyll, and are major factors controlling the distribution and abundance of seagrasses (Dennison et al. 1993, Kenworthy and Haunert 1991, Kenworthy and Fonseca 1996). Increases in color and turbidity values throughout the range of Johnson's seagrass are generally caused by high flows of freshwater discharged from water management canals, which can also reduce salinity. Wastewater and stormwater discharges, as well as from land runoff and subterranean sources, are also causes of increased turbidity. Degradation of water quality due to increased land use and poor water management practices continues to threaten the welfare of seagrass communities. Declines in water quality are likely to worsen, unless water management and land use practices can curb or eliminate freshwater discharges and minimize inputs of sediments and nutrients. A nutrient-rich environment caused by inorganic and organic nitrogen and phosphorous loading via urban and agricultural runoff stimulates increased algal growth that may smother or shade Johnson's seagrass, or shade rooted vegetation, and diminish the oxygen content of the water. Low oxygen conditions have a demonstrated negative impact on seagrasses and associated communities.

Based on a Trophic State Index of ambient water quality obtained in the northern and central region of Johnson's seagrass geographic range provided in a long-term monitoring program implemented by the SJRWMD, overall estuarine water quality was assessed as mostly good (67%) (Winkler and Ceric 2006). Only 28% of the stations sampled had fair water quality, while 6% had poor quality. 50% of the sampled estuarine sites were improving, while 6% were degrading, so many more sites were improving than were degrading. Forty-two percent of the lagoon sites had an insignificant trend while 3% had insufficient data to determine a trend. As water management experts have now become confident in the association between water quality and seagrass depth distribution, they have begun establishing water quality targets for the Indian River Lagoon based on seagrass as an indicator (Steward et al. 2005). There is a strong positive correlation between seagrass depth distribution and water quality which enables managers to predict where seagrasses will grow based on water quality and the availability of light. Given that at least half of the sampling stations were indicating long-term improvements in water quality, it can be assumed that seagrass abundance should not be negatively impacted if water and land use management programs continue to be effective. For example, carefully controlling or reducing water flows from discharge canals will moderate salinity fluctuations and reduce turbidity, color, and light attenuation values.

There has not been a comprehensive assessment of water quality published or reported for the southern range of Johnson's seagrass similar to the SJRWMD study. However, the South Florida Water Management District (SFWMD) is working to synthesize water quality information and to gain a more comprehensive understanding of the long-term status and trends of water quality in the southern range of Johnson's seagrass. Of particular concern is an assessment of the impacts of fluctuations in water quality corresponding with variation in climatology, especially "wet years" versus "dry years" variation. Future recovery efforts should include close coordination with the SFWMD and county environmental management agencies in Palm Beach and Dade counties to evaluate the status and trends of water quality in these regions of the species' distribution.

Here, we consider the possible effects of climate change (i.e., rising temperatures and sea levels) on seagrasses in general and on Johnson's seagrass in particular. The earth is projected to warm between 2°-4°C by 2100, and similar projections have been made for marine systems (Sheppard and Rioja-Nieto 2005). At the margins of temperate and tropical bioregions and within tidally-restricted areas where seagrasses are growing at their physiological limits, increased temperatures may result in losses of seagrasses and/or shifts in species composition (Short et al. 2007). The response of seagrasses to increased water temperatures will depend on the thermal tolerance of the different species and their optimum temperature for photosynthesis, respiration, and growth (Short and Neckles 1998). With future climate change and potentially warmer temperatures, there may be a 1-5 m rise in the seawater levels by 2100 when taking into account the thermal expansion of ocean water and melting of glaciers. Rising sea levels may adversely impact seagrass communities due to increases in water depths above present meadows reducing available light. Climate change may also reduce light by shifting weather patterns to cause increased cloudiness. Changing currents may cause erosion and increased turbidity and seawater intrusions higher up on land or into estuaries and rivers, which could increase landward seagrass colonization (Short and Neckles 1998). A landward migration of seagrasses with rising sea levels is a potential benefit, so long as suitable substrate is available for colonization.

It is uncertain how Johnson's seagrass will adapt to rising sea levels and temperatures. Much depends on how much temperatures increase and how quickly. For example, Johnson's seagrass that grows intertidally (e.g., in some parts of the Lake Worth Lagoon) may be affected by a slight change in temperature (since it may already be surviving under less than optimal conditions); however, this may be ameliorated with rising sea levels, assuming Johnson's seagrass would migrate landward with rising sea levels and assuming that suitable substrate would be available for a landward migration. However, rising sea levels could also adversely impact seagrass communities due to increases in water depths above present meadows reducing available light.

Reduction in light availability may benefit some seagrass species (e.g., *Halophila* species that require less light compared to the larger, canopy-forming species); therefore, much depends on the thermal tolerance of the different seagrass species and their optimum temperature for photosynthesis, respiration, and growth (Short and Neckles 1998). While sea level has changed many times during the evolutionary history of Johnson's seagrass, it is uncertain how this species will fare when considering the combined effects of rising temperatures and sea levels (in conjunction with other stressors, such as reduced salinity from freshwater runoff). It has been shown that evolutionary change in a species can occur within a few generations (Rice and Emery 2003), thus making it possible for seagrasses to cope if the changes occur at a rate slow enough to allow for adaptation.

Status Summary
Based on the results of 14 years of monitoring in the species' northern range (1994-2007) and 3 years of monitoring in the species' southern range (2006-2009), there has been no significant change in the northern or southern range limits of Johnson's seagrass (NMFS 2007). It appears that the populations in the northern range are stable and capable of sustaining themselves despite stochastic events related to severe storms (Steward et al. 2006) and fluctuating climatology. Longer-term monitoring data is needed to confirm the stability of the southern distribution of the species (NMFS 2007). Larger seagrasses, predominantly turtle grass (*Thalassia testudinum*), begin to out-compete Johnson's seagrass in this area. While there has been a slight extension in the known northern range (Virnstein and Hall 2009), the limits of the southern range appear to be stable (Latitude 25.75ºN in the vicinity of Virginia Key). There have been no reports of this species farther south of the currently known southern distribution.

As discussed in the Threats section, NMFS has determined that the most clearly identified threat to date is the possibility of mortality due to reduced salinity over long periods of time. The other potential threats discussed above (i.e., dredging/filling, construction and shading from in and over-water structures, propeller scarring and anchor mooring, trampling, storms, and siltation) were determined to be generally local and small-scale and are not considered threats to the survival and recovery of the species (NMFS 2007). It is uncertain how Johnson's seagrass and other seagrass species will fare due to the synergistic effects of rising temperatures and sea levels (in combination with other stressors, such as reduced salinity from freshwater runoff). It has been shown that evolutionary change in a species can occur within a few generations (Rice and Emery 2003), thus making it possible for seagrasses to cope if the changes occur at a rate slow enough to allow for adaptation.

5 Environmental Baseline

This section is a description of the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat (including designated critical habitat), and ecosystem, within the action area.[8] The environmental baseline is a "snapshot" of a species' health at a specified point in time. It does not include the effects of the action under review in the consultation.

By regulation, environmental baselines for biological opinions include the past and present impacts of all state, federal, or private actions and other human activities in the action area. We identify the anticipated impacts of all proposed federal projects in the specific action area of the consultation at issue, that have already undergone formal or early Section 7 consultation as well as the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02).

Focusing on the impacts of the activities in the action area specifically, allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals, and areas of designated critical habitat that occur in an action area, and that will be exposed to effects from the action under consultation. This is important because, in some phenotypic states or life history stages, listed individuals will commonly exhibit, or be more susceptible to, adverse responses to stressors than they would be in other states, stages, or areas within their distributions. The same is true for localized populations of endangered and threatened species: the consequences of changes in the fitness or performance of individuals on a population's status depends on the prior state of the population. Designated critical habitat is not different: under some ecological conditions, the physical and biotic features of critical habitat will exhibit responses that they would not exhibit in other conditions.

5.1 Sea Turtles

5.1.1 Status of Sea Turtles within the Action Area

Green and loggerhead sea turtles occur in the action area and may be adversely affected by the project. The action area does not include any nesting beach, important foraging habitat (e.g. nearshore hardbottom), or known breeding habitat. Sea turtles found in the immediate project area may travel widely throughout the Atlantic, Gulf of Mexico, and Caribbean Sea, and individuals found in the action area can potentially be affected by activities anywhere within this wide range. These impacts outside of the action area are discussed and incorporated as part of the overall status of the species as detailed in Section 3 above. Sea turtles that occur in the action area are highly migratory, as are all sea turtles species worldwide. For the species that are globally listed, the status of these species in the Atlantic (see Section 4) most accurately reflects the species' status within the action area. In Section 4, we presented available information on sea turtle population abundance and trends by species. The action area does not contain any important developmental habitat (e.g. nearshore hardbottom) and it is not near any nesting beaches.

5.1.2 Factors Affecting Sea Turtles in the Action Area

---

[8] The action area is defined by regulation as "all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action" (50 CFR 402.02).

NMFS has completed a number of Section 7 consultations to address the effects of federally-permitted fisheries and other federal actions on threatened and endangered sea turtle species, and when appropriate, has authorized the incidental taking of these species. Each of those consultations sought to minimize the adverse impacts of the action on sea turtles. NMFS has undertaken conservation actions under the ESA to address sea turtle takes in the fishing and shipping industries and other activities such as USACE dredging operations. The summary below of federal actions and the effects these actions have had or are having on sea turtles includes only those federal actions in, or with effects within, the action area that have already concluded or are currently undergoing formal Section 7 consultation.

Federal Vessel Activity and Operations
Potential sources of adverse effects from federal vessel operations in the action area include operations of the USN and USCG. NMFS has conducted formal consultations with the USCG and the USN on their vessel operations. Through the Section 7 process, where applicable, NMFS has and will continue to establish conservation measures for all these agency vessel operations to avoid or minimize adverse effects to listed species. Refer to the biological opinions for the USCG (NMFS 1995) and the USN (NMFS 1996, 1997a) for details on the scope of vessel operations for these agencies and conservation measures being implemented as standard operating procedures.

Dredging
The construction and maintenance of federal navigation channels and sand mining sites ("borrow areas") conducted by the USACE has been identified as a source of sea turtle mortality. Hopper dredges in the dredging mode are capable of moving relatively quickly, compared to sea turtle swimming speeds and can thus overtake, entrain, and kill sea turtles as the suction draghead of the advancing dredge overtakes the resting or swimming turtle. Entrained sea turtles rarely survive. NMFS completed a regional biological opinion on the impacts of USACE's South Atlantic coast hopper-dredging operations in 1997 for dredging in the USACE's South Atlantic Division (NMFS 1997b). The regional biological opinion on South Atlantic hopper dredging (SARBO) of navigational channels and borrow areas determined that hopper dredging would not adversely affect leatherback sea turtles in the South Atlantic Division (i.e., coastal states of North Carolina through Key West, Florida). The opinion did determine hopper dredging in the South Atlantic Division would adversely affect 4 sea turtle species (i.e., green, hawksbill, Kemp's ridley, and loggerheads) but would not jeopardize their continued existence. An ITS for those species was issued. Reinitiation of consultation on the SARBO has been triggered for a number of reasons, including listing of new species and designation of critical habitat that may be affected by these dredging activities.

ESA Permits
Sea turtles are the focus of research activities authorized by Section 10 permits under the ESA. Regulations developed under the ESA allow for the issuance of permits allowing take of certain ESA-listed species for the purposes of scientific research under Section 10(a)(1)(a) of the ESA. Authorized activities range from photographing, weighing, and tagging sea turtles incidentally taken in fisheries, to blood sampling, tissue sampling (biopsy), and performing laparoscopy on intentionally captured sea turtles. The number of authorized takes varies widely depending on the research and species involved, but may involve the taking of hundreds of sea turtles annually.

Most takes authorized under these permits are expected to be (and are) nonlethal, although lethal takes are sometimes authorized. Before any research permit is issued, the proposal must be reviewed under the permit regulations. In addition, since issuance of the permit is a federal activity, issuance of the permit by NMFS must also be reviewed for compliance with Section 7(a)(2) of the ESA to ensure that issuance of the permit does not result in jeopardy to the species or adverse modification of its critical habitat.

Federally-Managed Fisheries
Threatened and endangered sea turtles are adversely affected by fishing gears used throughout the continental shelf of the action area. Hook-and-line gear, trawl, and pot fisheries have all been documented as interacting with sea turtles.

For all fisheries for which there is a Fishery Management Plan (FMP) or for which any federal action is taken to manage that fishery, impacts have been evaluated under Section 7.

Finfish Fisheries
Adverse effects on threatened and endangered species from several types of fishing gear occur in the action area of the proposed action. Efforts to reduce the adverse effects of commercial fisheries are addressed through the ESA Section 7 process. Trawl, hook-and-line, gillnet, and cast net gear fisheries have all been documented as interacting with sea turtles. Several formal consultations have been conducted on the following fisheries that NMFS has determined are likely to adversely affect threatened and endangered species (including sea turtles): the South Atlantic and Gulf of Mexico coastal migratory pelagic fishery, and the Atlantic Highly Migratory Species shark fishery. An Incidental Take Statement (ITS) has been issued for interactions with sea turtles in each of these fisheries.

NMFS completed a Section 7 consultation on the continued authorization of the coastal migratory pelagic fishery in the South Atlantic (NMFS 2007c) where hook-and-line, gillnet, and cast net gears are used. The recreational sector uses hook-and- line gear. The hook-and-line effort is primarily trolling. The biological opinion concluded that green, hawksbill, Kemp’s ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery.

In 2012, NMFS issued a biological opinion on the continued authorization of Highly Migratory Species Atlantic shark fisheries (NMFS 2012). This commercial fishery uses bottom longline and gillnet gear. The recreational sector of the fishery uses only hook-and-line gear. To protect declining shark stocks, the proposed action seeks to greatly reduce the fishing effort in the commercial component of the fishery. These reductions are likely to greatly reduce the interactions between the commercial component of the fishery and sea turtles. The biological opinion concluded that green, hawksbill, Kemp’s ridley, leatherback, and loggerhead sea turtles may be adversely affected by operation of the fishery but that the proposed action was not expected to jeopardize the continued existence of any of these species.

Southeastern Shrimp Trawl Fisheries
Southeast U.S. shrimp fisheries target primarily brown, white, and pink shrimp in inland waters and estuaries through the state-regulated territorial seas and in federal waters of the EEZ. As sea turtles rest, forage, or swim on or near the bottom, they are captured by shrimp trawls pulled along the bottom. In 1990, the National Research Council (NRC) concluded that the Southeast shrimp trawl fisheries affected more sea turtles than all other activities combined and was the most significant anthropogenic source of sea turtle mortality in the U.S. waters, in part due to the high reproductive value of turtles taken in this fishery (NRC 1990).

On May 9, 2012, NMFS completed a Biological Opinion that analyzed the continued implementation of the sea turtle conservation regulations and the continued authorization of the Southeast U.S. shrimp fisheries in federal waters under the Magnuson-Stevens Act (NMFS 2012). The Opinion also considered a proposed amendment to the sea turtle conservation regulations that would withdraw the alternative tow time restriction at 50 CFR 223.206(d)(2)(ii)(A)(*3*) for skimmer trawls, pusher-head trawls, and wing nets (butterfly trawls) and instead require all of these vessels to use TEDs. The Opinion concluded that the proposed action would not jeopardize the continued existence of any sea turtle species. An ITS was provided that used trawl effort and capture rates as proxies for sea turtle take levels. The Biological Opinion requires NMFS to minimize the impacts of incidental takes through monitoring of shrimp effort and regulatory compliance levels, conducting TED training and outreach, and continuing to research the effects of shrimp trawling on listed species. Consultation for this fishery has recently been reinitiated.

Beach Nourishment
The USACE issues Clean Water Act permits for disposal of material in navigable waters of the United States, including beach nourishment. The activity of beach nourishment, especially when impacts include the loss of nearshore hardbottom habitat along the east coast of Florida, has been documented to result in injury and death of juvenile green sea turtles. Juvenile green turtles are known to utilize these high-energy, dynamic habitats for foraging and as refuge, and show a preference for this habitat even when abundant deeper-water sites are available. The loss of such limited habitat, especially when considering the cumulative loss as a result of beach nourishment activities occurring along the entire range of the habitat and continually over time, is expected to result in loss of foraging opportunities and protective refuge. The stresses are also expected to contribute to mortality of individuals already in poor condition as a result of disease or other factors (NMFS 2008a). Beach nourishment permitted by the USACE also often involves use of a hopper dredge to collect nourishment material, thus posing another route of adverse effects to sea turtles.

State or Private Actions
Maritime Industry
Private and commercial vessels, including fishing vessels, operating in the action area of this consultation also have the potential to interact with ESA-listed species. The effects of fishing vessels, recreational vessels, or other types of commercial vessels on listed species may involve disturbance or injury/mortality due to collisions or entanglement in anchor lines. Commercial traffic and recreational pursuits can also adversely affect sea turtles through propeller and boat strikes. The Sea Turtle Stranding and Salvage Network (STSSN) includes many records of

vessel interaction (propeller injury) with sea turtles where there are high levels of vessel traffic. The extent of the problem is difficult to assess because of not knowing whether the majority of sea turtles are struck pre- or post-mortem. It is important to note that minor vessel collisions may not kill an animal directly, but may weaken or otherwise affect it so it is more likely to become vulnerable to effects such as entanglements. NMFS and the USCG have completed several formal consultations on individual marine events that may affect sea turtles.

Coastal Development
Beachfront development, lighting, and beach erosion control all are ongoing activities along the Florida coastline. These activities potentially reduce or degrade sea turtle nesting habitats or interfere with hatchling movement to sea. Nocturnal human activities along nesting beaches may also discourage sea turtles from nesting sites. The extent to which these activities reduce sea turtle nesting and hatchling production is unknown. However, more and more coastal counties are adopting stringent protective measures to protect hatchling sea turtles from the disorienting effects of beach lighting.

State Fisheries
Commercial state fisheries are located in the nearshore habitat areas that comprise the action area. Recreational fishing from private vessels also occurs in the area. Observations of state recreational fisheries have shown that loggerhead sea turtles are known to bite baited hooks and frequently ingest the hooks. Hooked turtles have been reported by the public fishing from boats, piers, and beach, banks, and jetties and from commercial anglers fishing for reef fish and for sharks with both single rigs and bottom longlines (NMFS 2001). Additionally, lost fishing gear such as line cut after snagging on rocks, or discarded hooks and line, can also pose an entanglement threat to sea turtles in the area. A detailed summary of the known impacts of hook-and-line incidental captures to loggerhead sea turtles can be found in the TEWG reports (1998; 2000).

In August of 2007, NMFS issued a regulation (72 FR 43176, August 3, 2007) to require any fishing vessels subject to the jurisdiction of the United States to take observers upon NMFS's request. The purpose of this measure is to learn more about sea turtle interactions with fishing operations, to evaluate existing measures to reduce sea turtle takes, and to determine whether additional measures to address prohibited sea turtle takes may be necessary.

Other Potential Sources of Impacts in the Environmental Baseline
Marine Debris and Acoustic Impacts
A number of activities that may affect listed species in the action area of this consultation include anthropogenic marine debris and acoustic impacts. The impacts from these activities are difficult to measure. Where possible, conservation actions are being implemented to monitor or study impacts from these sources.

Marine Pollution and Environmental Contamination
Sources of pollutants along the coastal areas include atmospheric loading of pollutants such as polychlorinated biphenyls (PCBs), stormwater runoff from coastal towns and cities into rivers and canals emptying into bays and the ocean, and groundwater and other discharges (Carpenter et al, 1986). Nutrient loading from land-based sources such as coastal community discharges is

known to stimulate plankton blooms in closed or semi-closed estuarine systems (Bowen and Valiela, 2001; Rabalais 2002, Rabalais et al 2002). The effects on larger embayments are unknown. Although pathological effects of oil spills have been documented in laboratory studies of marine mammals and sea turtles (Vargo et al. 1986), the impacts of many other anthropogenic toxins have not been investigated.

Coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic can degrade marine habitats used by sea turtles (Colburn et al. 1996). The development of marinas and docks in inshore waters can negatively impact nearshore habitats. An increase in the number of docks built increases boat and vessel traffic. Fueling facilities at marinas can sometimes discharge oil, gas, and sewage into sensitive estuarine and coastal habitats. Although these contaminant concentrations do not likely affect the more pelagic waters, the species of turtles analyzed in this biological opinion travel between near shore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles.

There are studies on organic contaminants and trace metal accumulation in green and leatherback sea turtles (Aguirre et al. 1994; Caurant et al. 1999; Corsolini et al. 2000). Mckenzie et al. (1999) measured concentrations of chlorobiphenyls and organochlorine pesticides in sea turtle tissues collected from the Mediterranean (Cyprus, Greece) and European Atlantic waters (Scotland) between 1994 and 1996. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all the tissues sampled, including those from green and leatherback turtles (Storelli et al. 2008). Dietary preferences were likely the main differentiating factor among species. Decreasing lipid contaminant burdens with turtle size were observed in green turtles, most likely attributable to a change in diet with age. Sakai et al. (1995) found the presence of metal residues occurring in loggerhead turtle organs and eggs. Storelli et al. (1998) analyzed tissues from twelve loggerhead sea turtles stranded along the Adriatic Sea (Italy) and found that characteristically, mercury accumulates in sea turtle livers while cadmium accumulates in their kidneys, as has been reported for other marine organisms like dolphins, seals and porpoises (Law et al. 1991).

Conservation and Recovery Actions Benefiting Sea Turtles

NMFS has implemented a series of regulations aimed at reducing potential for incidental mortality of sea turtles from commercial fisheries in the action area. These include sea turtle release gear requirements for Atlantic HMS and Gulf of Mexico reef fish fisheries, and TED requirements for the southeastern shrimp fisheries. These regulations have relieved some of the pressure on sea turtle populations.

Under Section 6 of the ESA, NMFS may enter into cooperative research and conservation agreements with states to assist in recovery actions of listed species. NMFS has agreements with the state of Florida. Prior to issuance of these agreements, the proposal must be reviewed for compliance with Section 7 of the ESA.

Other Actions
A revised recovery plan for the loggerhead sea turtle was completed December 8, 2008 (NMFS and USFWS 2008). Recovery teams comprised of sea turtle experts have been convened and are currently working towards revising other plans based upon the latest and best available information. Five-year status reviews have recently been completed for green and loggerhead sea turtles. These reviews were conducted to comply with the ESA mandate for periodic evaluation of listed species to ensure that their threatened or endangered listing status remains accurate. Each review determined that no delisting or reclassification of a species status (i.e., threatened or endangered) was warranted at the time. However, further review of species data for the green sea turtles was recommended, to evaluate whether DPSs should be established for this species (NMFS and USFWS 2007).

Summary and Synthesis of Environmental Baseline for Sea Turtles
In summary, several factors adversely affect sea turtles in the action area. These factors are ongoing and are expected to occur contemporaneously with the proposed action. Fisheries in the action area likely had the greatest adverse impacts on sea turtles in the mid to late 80s, when effort in most fisheries was near or at peak levels. With the decline of the health of managed species, effort since that time has generally been declining. Over the past 5 years, the impacts associated with fisheries have also been reduced through the Section 7 consultation process and regulations implementing effective bycatch reduction strategies. However, interactions with commercial and recreational fishing gear are still ongoing and are expected to occur contemporaneously with the proposed action. Other environmental impacts including effects of vessel operations, additional military activities, dredging, oil and gas exploration, permits allowing take under the ESA, private vessel traffic, and marine pollution have also had and continue to have adverse effects on sea turtles in the action area in the past.

5.2 Corals

5.2.1 Status of Listed and Proposed Corals within the Action Area
In Section 4.2.2, we described the range-wide status of listed and proposed corals. Within the Broward County, staghorn coral occurs in some of the largest densities within the U.S. Recent surveys conducted by the National Coral Reef Institute have identified 35 dense patches of staghorn coral between Hollywood and Fort Lauderdale. Seven patches are near previously known existing locations and 28 newly identified areas. Initial approximations of areal coverage suggest the sites totaled over 110,000 m^2 of previously unknown dense patches of staghorn coral. These new discoveries have the potential of more than tripling the area of previously documented staghorn coral (B. Walker, National Coral Reef Institute, pers. comm. to J. Karazsia, NMFS, October 21, 2013).

Within Broward County, all of the proposed corals occur in varying, but relatively low densities (Gilliam 2011). Recent surveys adjacent to the Port Everglades expansion indicate that 6 of the proposed corals as well as staghorn coral are present nearby the action area, see Table 3 (Gilliam and Walker 2011). Within the Port Everglades expansion area, knobby star coral, mountainous star coral, lobed star coral, elliptical star coral, rough cactus coral, and Lamarck's sheet coral occurs on the middle reef tract and outer reef tract adjacent to the channel and within the proposed extension and flare area.

Table 3. Summary Data for Staghorn and Proposed Coral Species Adjacent to Port Everglades (Gilliam and Walker 2011)

Species Name	Number of Colonies	Density (colonies/acre)
Acropora cervicornis	823	1.12
Agaricia lamarcki	912	1.24
Dichocoenia stokesii	376	0.51
Orbicella annularis	262	0.36
Orbicella faveolata	4030	5.48
Orbicella franksi	298	0.41
Mycetophyllia ferox	26	0.04

5.2.2 Factors Affecting Listed and Proposed Corals within the Action Area

Coral colonies are non-motile and susceptible to relatively localized adverse effects as a result. Localized adverse effects to listed and proposed corals in the action area are likely from many of the same stressors affecting these species throughout their range, namely ocean warming, ocean acidification, disease, anthropogenic breakage and intense weather events (i.e., hurricanes and extreme cold water disturbances). NMFS has completed a number of Section 7 consultations to address the effects of federal actions on staghorn corals, and when appropriate, has authorized the incidental taking of this species. Each of those consultations sought to minimize the adverse impacts of the action on staghorn coral. The summary below of federal actions and the effects of these actions includes only those federal actions in, or with effects within, the action area that have already concluded or are currently undergoing formal Section 7 consultation.

Federal Actions

Federal actions that may adversely affect listed and proposed corals in the action area include:

- **Commercial and recreational fisheries authorized by the National Marine Fisheries Service.** Certain types of fishing gear (e.g., hook-and-line, trap gear, nets) may adversely affect coral species. NMFS previously completed a biological opinion evaluating the impacts of Gulf of Mexico/South Atlantic spiny lobster fishery on *A. cervicornis*. The opinion concluded trap gear used in the fishery may adversely affect *A. cervicornis* corals via fragmentation/breakage and abrasion (primarily from storm mobilized trap gear), but those effects were not likely to jeopardize the species continued existence. NMFS is continuing to collect data to analyze the impacts of federal fisheries and will conduct ESA Section 7 consultations as appropriate.

EPA and USACE-permitted discharges to surface waters and dredge-and-fill. Shoreline and riparian disturbances (whether in the riverine, estuarine, marine, or floodplain environment) resulting in discharges may retard or prevent the reproduction, settlement, reattachment, and development of listed or proposed corals (e.g., land development and runoff, and dredging and disposal activities, result in direct deposition of sediment on corals, shading, and lost substrate for fragment reattachment or larval settlement). These activities can directly affect *A.*

cervicornis via fragmentation/breakage or abrasion. The activities may also affect listed and proposed coral species by physically altering or removing benthic habitat suitable for colonization. Dredge-and-fill activities may also cause increases in sedimentation that may cause shading, deposition of sediment onto coral colonies, and/or loss of substrate for fragment reattachment or larval settlement. The 1997 RBO is currently undergoing a reinitiation of consultation due to the listing of *A. cervicornis* and *A. palmata,* among other things.

- **EPA-regulated discharge of pollutants, such as oil, toxic chemicals, radioactivity, carcinogens, mutagens, teratogens, or organic nutrient-laden water, including sewage water, into the waters of the United States.** Elevated discharge levels may cause direct mortality, reduced fitness, or habitat destruction/modification. The EPA has been involved in ongoing litigation over the sufficiency of standards promulgated by the State of Florida to regulate discharges of nutrients into state waters, including habitats occupied by the listed and proposed corals. NMFS is engaged in consultation with the EPA regarding their approval of the state's standards.

- **Coral Nurseries.** NMFS has issued 3 separate biological opinions for the establishment of staghorn coral nurseries and restoration projects within Broward County (one to Biscayne National Park, one to NMFS Habitat Conservation/Restoration Center, and one to The Nature Conservancy). The activities include collecting coral fragments and growing them within nurseries and then outplanting them onto the natural reefs. In all cases NMFS has determined that the nursery and restoration activities would not jeopardize the continued existence of staghorn corals.

Other Non-Federal Actions Affecting Listed and Proposed Corals.
Poor boating and anchoring practices, as well as poor diving and snorkeling techniques cause abrasion and breakage of *Acropora cervicornis*. Commercial and recreational vessel traffic can adversely affect listed and proposed corals through propeller scarring, propeller wash, and accidental groundings. Anthropogenic sources of marine pollution, while difficult to attribute to a specific federal, state, local or private action, may indirectly affect corals in the action area. Sources of pollutants in the action area include atmospheric loading of pollutants such as PCBs, storm water runoff from coastal towns, and runoff into canals and rivers that empty into bays and groundwater. Nutrients, contaminants, and sediment from point and non-point sources cause direct mortality and the breakdown of normal physiological processes. Additionally, these stressors create an unfavorable environment for reproduction and growth.

Nutrient loading from land-based sources, such as coastal communities and agricultural operations, are known to have adverse effects on corals. Lapointe et al. (2004) directly linked wastewater discharges in the Florida Keys with adverse effects to the nearby coral reef communities. Within the past 6 years, offshore wastewater outfalls in Broward County have been decommissioned, as part of implementation of Chapter 2008-232, Laws of Florida, which prohibits the construction of new domestic wastewater ocean outfalls, sets out a timeline for the elimination of existing domestic wastewater ocean outfalls by 2025, and requires that a majority of the wastewater previously discharged be beneficially reused. This law was enacted in part because of the adverse effects of effluent to corals.

Diseases have been identified as a major cause of coral decline. Although the most severe mortality resulted from an outbreak in the early 1980s, diseases (i.e., white band disease) are still present in *Acropora cervicornis* populations and continue to cause mortality.

Hurricanes and large coastal storms could also significantly harm *Acropora cervicornis*. Due to its branching morphology, it is especially susceptible to breakage from extreme wave action and storm surges. Historically, large storms potentially resulted in an asexual reproductive event, if the fragments encountered suitable substrate, attached, and grew into a new colony. However, in the recent past, the amount of suitable substrate is significantly reduced; therefore, many fragments created by storms die. Hurricanes are also sometimes beneficial, if they do not result in heavy storm surge, during years with high sea surface temperatures, as they lower the temperatures providing fast relief to corals during periods of high thermal stress (Heron et al. 2008). However, major hurricanes have caused significant losses in coral cover and changes in the physical structure of many reefs. According to the NOAA Historical Hurricane Tracks website, approximately, 29 hurricanes or tropical storms have impacted the area within 20 nautical miles of Fort Lauderdale, since records have been kept (1859-2013).

Several types of fishing gears used within the action area may adversely affect listed and proposed corals. Longline, other types of hook-and-line gear, and traps have all been documented as interacting with corals in general, though no data specific to listed corals are available. Available information suggests hooks and lines can become entangled in reefs, resulting in breakage and abrasion of corals. Traps have been found to be the most damaging; lost traps and illegal traps were found to result in greater impact to coral habitat because they cause continuous habitat damage until they degrade.

Conservation and Recovery Actions Benefiting Listed Corals
Research, restoration, and education and outreach activities, as part of the NMFS's ESA program, as well as through NOAA's Coral Reef Conservation Program (CRCP), are ongoing through the southeast region. NOAA's Restoration Center and state and territorial partners conduct grounding response and restoration activities throughout the U.S. jurisdictions. The summaries below discuss these measures in more detail.

Regulations Reducing Threats to Listed Corals
Numerous management mechanisms exist to protect corals or coral reefs in general. Prior to the ESA listing of elkhorn and staghorn corals, federal regulatory mechanisms and conservation initiatives most beneficial to branching corals have focused on addressing physical impacts, including damage from fishing gear, anchoring, and vessel groundings. NMFS has implemented a Section 4(d) rule to establish "take" prohibitions for listed corals. Such regulations are determined to be necessary and advisable to provide for the conservation of threatened species, and may prohibit many actions automatically prohibited for endangered species, including but not limited to: importing or exporting species from or into the United States; taking of species from U.S. waters, its territorial sea, or the high seas; or possessing or selling species. On October 29, 2008, NMFS published a final Section 4(d) rule extending all the Section 9 take prohibitions to listed elkhorn and staghorn corals. These prohibitions include the import, export, or take of elkhorn or staghorn corals for any purpose, including commercial activities. The 4(d)

rule for listed *Acropora* has exceptions for some activities, including scientific research and species enhancement, and restoration carried out by authorized personnel.

In addition, the Coral Reef Conservation Act and the two Magnuson-Stevens Act Coral and Reef Fish Fishery Management Plans (Caribbean) require the protection of corals and prohibit the collection of hard corals. Depending on the specifics of zoning plans and regulations, marine protected areas (MPAs) can help prevent damage from collection, fishing gear, groundings, and anchoring.

The State of Florida regulates activities that involve and occur in coral reefs in Florida. Statutes and rules protect all corals from collection, commercial exploitation, and injury/destruction on the sea floor (FS 253.001, 253.04, Chapter 68B-42.008 and 68B-42.009), except as authorized by a Special Activity License for the purposed of research. Additionally, Florida has a comprehensive state regulatory program that regulates most land, including upland, wetland, and surface water alterations throughout the state.

Other Listed Coral Conservation Efforts

Recovery Planning and Implementation

A draft recovery plan for elkhorn and staghorn corals is required by a settlement agreement to be published no later than September 7, 2014. The recovery team is comprised of fishers, scientists, managers, and agency personnel from Florida, Puerto Rico, and U.S.V.I., and federal representatives. Similar plans will be identified for proposed coral species should the listings become finalized.

Even in the absence of a recovery plan, NMFS and its partners have implemented numerous recovery actions since the time of listing, consistent with NMFS's Recovery Outline for elkhorn and staghorn corals. Generally, these activities fall into the following categories:

- Monitoring and mapping
- Life history, disease, and threat impact research
- In-situ and ex-situ propagation and outplanting
- Reduction of and restoration of impacts from physical disturbances
- Reduction of impacts from land-based sources of pollution
- Outreach and education

Summary and Synthesis of Environmental Baseline for Listed and Proposed Corals

In summary, several factors are presently adversely affecting listed and proposed corals within the action area. These factors are ongoing and are expected to occur contemporaneously with the proposed action:

- Disease outbreaks
- Temperature-induced bleaching events
- Ocean acidification
- Major storm events
- Upland and coastal activities that will continue to degrade water quality and decrease water clarity necessary for coral growth
- Dredge-and-fill activities

- Interaction with fishing gear and adverse effects of fishing
- Vessel traffic that will continue to result in abrasion and breakage due to accidental groundings and poor anchoring techniques
- Poor diving and snorkeling techniques that will continue to abrade and break corals

These activities are expected to combine to adversely affect the recovery of staghorn and proposed corals throughout their ranges, and in the action area.

5.3 Status of Elkhorn and Staghorn Coral Designated Critical Habitat within the Action Area

In Section 4.2.6, we described the range-wide status of designated *Acropora* critical habitat. In summary, the Florida area of *Acropora* spp. critical habitat comprises approximately 1,329 square miles (3,442 sq km) of marine habitat offshore of Palm Beach, Broward, Miami-Dade, and Monroe counties, Florida, and encompasses the entire Florida Reef Tract beginning east of Palm Beach County and extending south along the Florida Keys. Within the action area, there are approximately 19,200 acres (~30 square miles) of designated critical habitat, which includes both the areas affected by the Port expansion and the areas associated with the blended mitigation plan (discussed in Consultation History section of this Opinion) in which the nurseries and outplanting sites will occur.

Factors Affecting Critical Habitat within the Action Area

Localized adverse effects to designated critical habitat in the action area are likely from many of the same stressors affecting the critical habitat throughout their range, namely activities that may increase turf- or macroalgal cover (i.e., releases of nutrients or reduction in herbivory) or increase sediment cover.

Federal Actions

Numerous activities funded, authorized, or carried out by federal agencies have been identified as threats and may affect elkhorn and staghorn corals' critical habitat in the action area. To date, however, few consultations on activities affecting critical habitat within the action area have been completed.

- **USACE-permitted dredge-and-fill activities.** The activities may impact critical habitat by physically altering or removing benthic habitat suitable for colonization. Dredge-and-fill activities may also cause increases in sedimentation that may cause loss of substrate for fragment reattachment or larval settlement. The 1997 RBO on navigation channel maintenance using hopper dredges is currently undergoing a reinitiation of consultation, to address the impacts of these activities on coral critical habitat among other things, and will evaluate the effects of certain dredge-and-fill activities that occur within the action area. In the past century, 3 major ports have been constructed in southeast Florida. A total of approximately 772 acres of coral reef habitat has been impacted via direct removal and burial (Walker et al. 2012a). Several beach renourishment projects have been completed in Broward County. In 2006, Segment III renourishment project resulted in over 43 acres of nearshore reef impacts via sediment burial (Prekel et al. 2008).

- **EPA-regulated discharge of pollutants, such as oil, toxic chemicals, radioactivity, carcinogens, mutagens, teratogens, or organic nutrient-laden water, including sewage water, into the waters of the United States.** Elevated nutrients can lead to increased algal growth. The EPA has been involved in ongoing litigation over the sufficiency of standards promulgated by the State of Florida to regulate discharges of nutrients into state waters, including habitats occupied by the listed and proposed corals. NMFS is engaged in consultation with the EPA regarding their approval of the state's standards.

Other Non-Federal Actions Affecting Elkhorn and Staghorn Critical Habitat.
The State of Florida regulates activities that involve and occur in coral reefs in Florida. Statutes and rules protect all corals from collection, commercial exploitation, and injury/destruction on the seafloor (FS 253.001, 253.04, Chapter 68B-42.008 and 68B-42.009), except as authorized by a Special Activity License for the purposed of research. Therefore, the State regulates alterations to the reef. Additionally, Florida has a comprehensive state regulatory program that regulates most land, including upland, wetland, and surface water alterations throughout the state, resulting in regulation of land-based sources of nutrients or sediment that may adversely affect *Acropora* critical habitat.

Vessel groundings and anchor damage from commercial and recreational vessels within southeast Florida have historically resulted in severe negative impacts to the Florida Reef Tract. According to Sansgaard (2013) the Florida Department of Environmental Protection's (FDEP) Coral Reef Conservation Program (CRCP) has responded to, and managed, 124 of incidents related to vessel groundings and anchor damage. Typically only large vessel groundings alter the substrate to render it unconsolidated. However, several of the documented events have been large vessels. For example, in 2006, the M/V Clipper Lasco (a 645-ft cargo ship) grounded offshore of Fort Lauderdale resulting in over 6,000 square feet (ft^2) of reef impacted. However, due to the large number of vessel groundings in the area, the U.S. Coast Guard relocated the anchorage and no large vessel groundings have occurred since 2009.

Conservation and Recovery Actions Benefiting Coral Critical Habitat in the Action Area
The NOAA Coral Reef Conservation Program provides funding for several activities with an education and outreach component for informing the public about the importance of the coral reef ecosystem and the status of listed corals. The Southeast Regional Office of NMFS has also developed outreach materials regarding the listing of elkhorn and staghorn corals, the Section 4(d) regulations, and the designation of critical habitat. These materials have been circulated to constituents during education and outreach activities and public meetings, and as part of other Section 7 consultations, and are readily available on the website: http://sero.nmfs.noaa.gov/pr/esa/acropora.htm.

Numerous management mechanisms exist to protect corals and the habitats on which they grow, thus indirectly benefiting *Acropora* designated critical habitat. The Coral Reef Conservation Act and the two Coral and Coral Reef Fishery Management Plans under the Magnuson-Stevens Act require the protection of corals and prohibit the collection of hard corals. Depending on the specifics of zoning plans and regulations, marine protected areas (MPAs) can help prevent damage from collection, fishing gear, groundings, and anchoring; however, no MPAs occur within the action area.

5.4 Johnson's Seagrass

5.4.1 Status of Johnson's Seagrass within the Action Area

Based on the results of the southern transect sampling, it appears there is a relatively continuous, although patchy, distribution of the species from Jupiter Inlet to Virginia Key, at least during periods of relatively good environmental conditions and no significant large-scale disturbances (NMFS 2007).

The project area includes several small patches of Johnson's seagrass, mostly intermixed with other seagrass species. The majority of the seagrass found in the action area will not be affected by the project.

5.4.2 Factors Affecting Johnson's Seagrass within the Action Area

A wide range of activities funded, authorized, or carried out by federal agencies may affect the essential habitat requirements of Johnson's seagrass.

Federal Actions

- **Dock/Marina Construction, boat shows, bridge/highway construction, residential construction, and shoreline stabilization.** NMFS has consulted on numerous projects in or near the action area that have adversely affected Johnson's seagrass. The majority of these projects were single- or multi-family dock construction that resulted in a few hundred square feet of impacts to Johnson's seagrass. However, a few projects resulted in more significant impacts. Newer construction is encouraged to follow the NMFS-USACE dock construction guidelines and the Johnson's Seagrass Key in order to minimize shading impacts to Johnson's seagrass. NMFS and the USACE have covered many of the impacts to Johnson's seagrass in several programmatic biological opinions on regional general permitting activities, which ensure that issuance of the general permits as a whole are not likely to jeopardize the affected species.

- **EPA and the USACE permitted freshwater discharges into waterways**. Freshwater discharges can alter the salinity essential feature for Johnson's seagrass. Water quality and transparency within the range of Johnson's seagrass are affected by storm water and agricultural runoff, wastewater discharges, and other point and non-point source discharges. The most clearly identified and manageable threat to the survival and recovery of Johnson's seagrass is the possibility of mortality due to reduced salinity over long periods of time (NMFS 2007). High-volume freshwater discharges from Lake Okeechobee flow downstream to the mouth of the St. Lucie River and have the potential to adversely affect Johnson's seagrass. NMFS recently completed consultation with the USACE on the programmatic impacts of the Comprehensive Everglades Restoration Plan (CERP), which may help to alleviate the frequency of high-volume freshwater discharges from Lake Okeechobee to Johnson's seagrass habitats.

Other Non-Federal Actions Affecting Johnson's seagrass

Natural Disturbances

Large-scale weather events, such as tropical storms and hurricanes, while they often generate runoff conditions that decrease water quality, also produce conditions (wind setup and abrupt water elevation changes) that can increase flushing rates. The effects of storms can be complex. Specifically documented storm effects on healthy seagrass meadows have been relatively minor and include: (1) scouring and erosion of sediments; (2) erosion of seeds and plants by waves, currents, and surge; (3) burial by shifting sand; (4) turbidity; and (5) discharge of freshwater, including inorganic and organic constituents in the effluents (Oppenheimer 1963, van Tussenbroek 1994, Whitfield et al. 2002, Steward et al. 2006). Storm effects may be chronic, e.g., due to seasonal weather cycles, or acute, such as the effects of strong thunderstorms or tropical cyclones. Studies have demonstrated that healthy, intact seagrass meadows are generally resistant to physical degradation from severe storms, whereas damaged seagrass beds may not be as resilient (Fonseca et al. 2000, Whitfield et al. 2002). In the late summer and early fall of 2004, 4 hurricanes passed directly over the northern range of Johnson's seagrass in the Indian River Lagoon. A post-hurricane random survey in the area of the Indian River Lagoon affected by the 4 hurricanes indicated the presence of Johnson's seagrass was similar to that reported by the SJRWMD transect surveys prior to the storms. This indicates that while the species may temporarily decline, under the right conditions it can recover quickly (Virnstein and Morris 2007). Furthermore, despite evidence of longer-term reductions in salinity, increased water turbidity, and increased water color associated with higher than average precipitation in the spring of 2005, there was no evidence of long-term chronic impacts to seagrasses and no direct evidence of damage to Johnson's seagrass that could be considered a threat to the survival of the species (Steward et al. 2006).

State and Federal Activities That May Benefit Johnson's Seagrass

State and federal conservation measures exist to protect Johnson's seagrass and its habitat under an umbrella of management and conservation programs that address seagrasses in general (Kenworthy et al. 2006). These conservation measures must be continually monitored and assessed to determine if they will ensure the long-term protection of the species and the maintenance of environmental conditions suitable for its continued existence throughout its geographic distribution.

5.5 Summary and Synthesis of Environmental Baseline

In summary, several factors are presently adversely affecting green and loggerhead sea turtles, Johnson's seagrass, listed and proposed for listing corals, and designated critical habitat for elkhorn and staghorn corals in the action area. These factors are ongoing and are expected to occur contemporaneously with the proposed action:

- Interaction with commercial and recreational fishing gear
- Dredge-and-fill activities, including channel dredging and beach re-nourishment/restoration activities
- Runoff containing toxins and pollutants from land-based sources
- Disease outbreaks
- Major storm events
- Upland and coastal activities will continue to degrade water quality and decrease water clarity necessary for coral growth

- Poor vessel anchoring as well as poor diving and snorkeling techniques will continue to abrade and break corals

These activities are expected to combine to adversely affect the recovery of green and loggerhead sea turtles, Johnson's seagrass, and proposed and listed corals throughout their ranges, and in the action area.

6 Effects of the Action

As described below, NMFS believes that the proposed action may adversely affect loggerhead and green sea turtles, Johnson's seagrass, staghorn coral and corals proposed for listing under the ESA, and designated critical habitat for staghorn coral. Because the action will result in adverse effects to these species, we must evaluate whether the action is likely to jeopardize the continued existence of any of these species or likely to cause destruction or adverse modification to critical habitat.

6.1 Effects of the Action on Sea Turtles

In Section 3, we determined listed species of sea turtles likely to be adversely affected via any or all portions of the proposed action include green and loggerhead sea turtles. Potential routes of adverse effects of the proposed action on sea turtles are limited to hopper dredging.

Previous NMFS biological opinions have determined that hopper dredges may adversely affect loggerhead and green sea turtles through entrainment by the draghead. Hopper dredges will only be used to suction off accumulated shoal material from the existing. This may take anywhere from a few days to a few weeks depending on the amount of material that has shoaled into the entrance channel. Between 2005 and 2013 approximately 100,000 cy of material shoaled in the Port Everglades entrance channel (pers. comm. Terri Jordan-Sellers, USACE, to K. Logan, NMFS, February 2014). Assuming a similar amount of shoal material is to be removed by hopper dredge and assuming that the contractor uses a smaller, 3,000-cy-capacity hopper dredge (with an average load value of 2,500 cy), they would need to complete approximately 40 trips total to the ODMDS.

During dredging operations, protected species observers will live aboard the dredge, monitoring every load, 24 hours a day, for evidence of dredge-related impacts to protected species, particularly sea turtles. Observers will also maintain a bridge watch for protected species and keep a logbook noting the date, time, location, species, number of animals, distance and bearing from dredge, direction of travel, and other information, for all sightings. During all phases of dredging operations, the dredge and crew will be required to adhere to NMFS's *Sea Turtle and Smalltooth Sawfish Construction Conditions*.

Since there has never been a reported sea turtle take from hopper dredging in Port Everglades, Port of Miami, or Key West, we relied on data from the nearest harbor with reported takes, Palm Beach Harbor, in order to estimate potential take by hopper dredges in the action area during the proposed 5-year dredging action. From 1994 through 2011, hopper dredging of the Palm Beach Harbor generated approximately 2,446,916 cy of material (Table 4). Eleven sea turtles were documented/observed as taken in hopper dredges during these dredging events. This equates to a catch per unit effort (CPUE) of 0.00000449 turtles per cubic yard dredged.

Table 4. Dredged Material Removed and Sea Turtle Takes During Dredging in the Palm Beach Harbor, 1994-2011 (USACE Sea Turtle Data Warehouse 2014)

Year	Quantity of Dredged Material (Cubic yards) Palm Beach Harbor	Loggerhead	Green	Total Turtles
1994	181,338			
1995	179,330	3	2	5
1996	154,847	1	1	2
1997	219,177			
1998	73,349			
1999	64,779			
2000	187,340	1		1
2001	112,446			
2002	184,935			
2003	111,625	1		1
2004	343,770	1		1
2005	318,874	1		1
2006	70,698			
2007	12,000			
2008	157,828			
2009	43,735			
2010	64,068			
2011	66,777			
Total	2,446,916	8	3	**11**
CPUE	0.00000449			

Using this data we can calculate that the proposed project will take 0.45 turtles (0.00000449 X 100,000 = 0.45), rounded up to 1 turtle.

NMFS has previously determined that dredged material screening is only partially effective at detecting entrained turtles, and observed interactions likely provide only partial estimates of total sea turtle mortality. NMFS believes that some turtles killed by hopper dredges go undetected because body parts are forced through the sampling screens by water pressure and are buried in the dredged material, or animals are crushed or killed but their bodies or body parts are not entrained by the suction and so the interactions may go unnoticed. Mortalities are only noticed and documented when body parts float, are large enough to be caught in the screens, and can be identified as sea turtle parts. Body parts that are forced through the suction dragheads' 4-inch (or greater) inflow screens by the suction-pump pressure and that do not float are very unlikely to be observed, since they will sink to the bottom of the hopper and not be detected by the overflow screening.

Unobserved interactions are not documented, thus, observed interactions may under-represent actual lethal interactions. There may have been unobserved takes in previous dredging operations at Port Everglades.

It is not known how many turtles are killed but unobserved. Thus, to be conservative, in the 1993 Regional Biological Opinion on hopper dredging issued to the U.S. Army Corps of Engineers for their Gulf of Mexico District's (i.e., Jacksonville, Mobile, New Orleans, and Galveston) maintenance dredging and beach renourishment operations, NMFS estimated that up to 1 out of 2 impacted turtles may go undetected (i.e., that observed interactions constitute only 50% of total takes). We will apply this longstanding conservative assumption in the present opinion, since we have no new information that would change the basis of that previous conclusion and estimate. Therefore, our jeopardy analysis will account for total takes (observed takes plus undetected takes). Our Incidental Take Statement (ITS) is based on observed takes, not only because observed mortality gives us an estimate of unobserved mortality, but because observed, documented take numbers serve as triggers for some of the reasonable and prudent measures, and for potential reinitiation of consultation if actual observed takes exceed the anticipated/authorized number of observed takes.

Experience has shown that the vast majority of hopper-dredge impacted turtles are immediately killed by being crushed or through dismemberment from being trapped underneath and rolled under the heavy suction dragheads and/or by the violent forces they are subjected to during entrainment through the dredges' powerful, high-velocity dredge pumps. A very few turtles (over the years, a fraction of a percent) survive entrainment in hopper dredges, usually smaller juveniles that are sucked through the pumps without being dismembered or badly injured. Often they will appear uninjured only to die days later of unknown internal injuries, while in rehabilitation. Therefore, we are conservatively predicting that all takes by hopper dredges will be lethal.

As discussed above, NMFS estimates that there will be 2 incidental, lethal interactions (1 observed and 1 unobserved). Because more loggerheads were taken than greens in dredging activity in Palm Beach Harbor (approximately 2.5 times as many), we anticipate that the turtles taken will be loggerheads, but we cannot rule out that greens may be taken. Green sea turtles made up 27% of entrainments at Palm Beach Harbor hopper dredging. Given the growth of the green sea turtle population over the past decade and increased nesting of greens on Florida beaches, we believe green sea turtles are relatively more abundant in nearshore Florida waters than previously (see Figure 7). By comparison, the loggerhead population has not enjoyed the same rate of long-term increase (see Figure 6). Therefore, we believe that the observed take might well consist of 1 green or 1 loggerhead, and, for the purposes of this Opinion, that is our anticipated observed take by species. However, to be most conservative, in our jeopardy analysis, we will assume that both takes will occur to just reproductively mature females of just one species, i.e., that 2 loggerheads or 2 greens will be lethally taken.

6.2 Effects of the Action on Johnson's Seagrass

NMFS believes the proposed action is likely to adversely affect Johnson's seagrass, which is listed as threatened under the ESA. The ESA expressly provides only limited prohibitions on take of endangered plants (*See* ESA section 9(a)(2), 16 U.S.C. § 1538(a)(2)), and NMFS has not promulgated any 4(d) rule for Johnson's seagrass. Thus, take of Johnson's seagrass resulting from the proposed action is not prohibited, and no incidental take statement or reasonable and prudent measures will be issued. However, because the action will result in adverse effects to Johnson's seagrass, we must evaluate whether the action is likely to jeopardize the continued existence of the species.

Johnson's seagrass will be directly removed via dredging; no other types of effects, such as sedimentation, are expected to impact this species. Utilizing data from surveys conducted by Dial Cordy, Inc., in 2000, 2006, and 2009, we performed an independent GIS analysis to determine cumulative coverage of Johnson's seagrass. This approach is consistent with the methodology used by NOAA's Habitat Conservation office in determining seagrass impacts for this project. We determined that 4.67 acres of Johnson's seagrass will be permanently removed via dredging (see Table 5 and Figure 11).

Table 5. Cumulative Coverage of Johnson's Seagrass

	Cumulative Average Coverage (acres)
Johnson's Seagrass	4.379
Mixed Seagrass*	0.289
Total	4.668

*Mixed seagrass beds were calculated assuming 50% coverage of Johnson's seagrass. Transect data indicated a range of coverages from less than 1% to approximately 50%; therefore, to be conservative, we will use 50% for all the mixed beds.

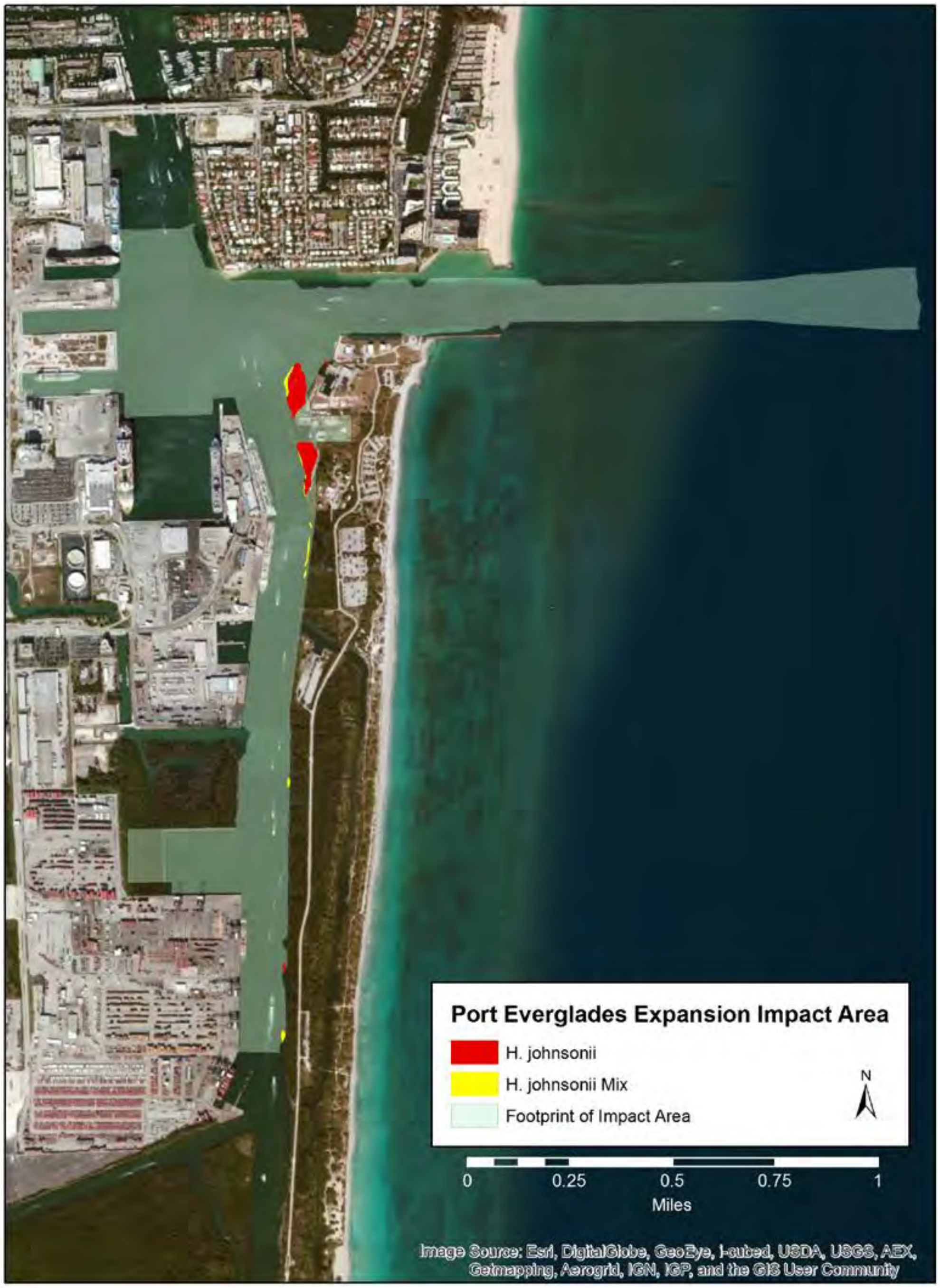

Figure 11. Johnson's seagrass within the project area

6.3 Effects of the Action on Coral Designated Critical Habitat

As described below, NMFS believes the proposed action will both adversely affect and benefit designated critical habitat for staghorn coral. The Florida area, which will be affected by the proposed action, comprises approximately 1,329 square miles of listed coral critical habitat. The physical feature essential to the conservation of staghorn coral is defined as substrate of suitable quality and availability, in water depths from mean high water to 30 m, to support larval settlement and recruitment, and reattachment of asexual fragments. Substrate of suitable quality and availability is defined as natural consolidated hardbottom or dead coral skeleton that is free from turf or fleshy macroalgae cover and sediment cover. We used hardbottom mapping data for south Florida (Walker et al., 2008b) to determine the amount of the critical habitat essential feature that could be affected by the project. Approximately 139 acres of coral critical habitat will be adversely affected by the project. Additionally, approximately 22 acres of reef will be populated with dense stands of staghorn coral as part of the blended mitigation plan, accelerating the conservation function of these areas of coral critical habitat. Based on these adverse and beneficial effects to critical habitat, we must evaluate whether the proposed action may result in the destruction or adverse modification of critical habitat; if so, NMFS must develop reasonable and prudent alternatives to avoid such impacts.

The Port Everglades Expansion project includes various types of impacts to coral reef and hardbottom habitats through directly dredging or blasting, anchoring and cable dragging, and sedimentation. To determine the nature and extent of impacts to coral critical habitat from the proposed action, we used Figure 12 below which has been adapted from Walker et al. (2008b) and includes the project boundaries (black lines) overlaid on the benthic habitat map produced by Dr. Walker (colored areas). The figure shows the dredge footprint (inner black lines) and the adjacent 150 meter area (outer black line). The area at the end of the channel (in yellow) includes the 6.11 acre area below the -57 ft dredge depth where we believe that fracturing and other impacts will occur from removing the reef structure above this depth. Hardbottom habitat types are identified and color coded, sand areas are indicated in grey. As indicated in Table 6, below, we believe that there will be permanent impacts from dredging and blasting to the habitat areas within the dredge footprint (channel) and sedimentation impacts (both permanent and temporary) to the area within 150 meters adjacent to the channel. Additionally, we believe there will be permanent impacts (fracturing, etc.) to the hardbottom area located along the outer reef tract, below the -57 ft dredge depth. Furthermore, there may be some additional anchor and cable drag impacts (potential impacts) to 19.31 acres of habitat within the 150 meters adjacent to the channel in the event that the USACE selects a contractor that will need to anchor outside of the channel[9].

[9] At this time the mitigation plan and incremental cost analysis is in draft form and may contain different inputs than what is analyzed in this Opinion. Impact estimates used in this Opinion are the most conservative to be consistent with the requirements of the ESA. Any changes made to the mitigation plan as a result of inputs used will not result in less than 38,254 staghorn colony outplants.

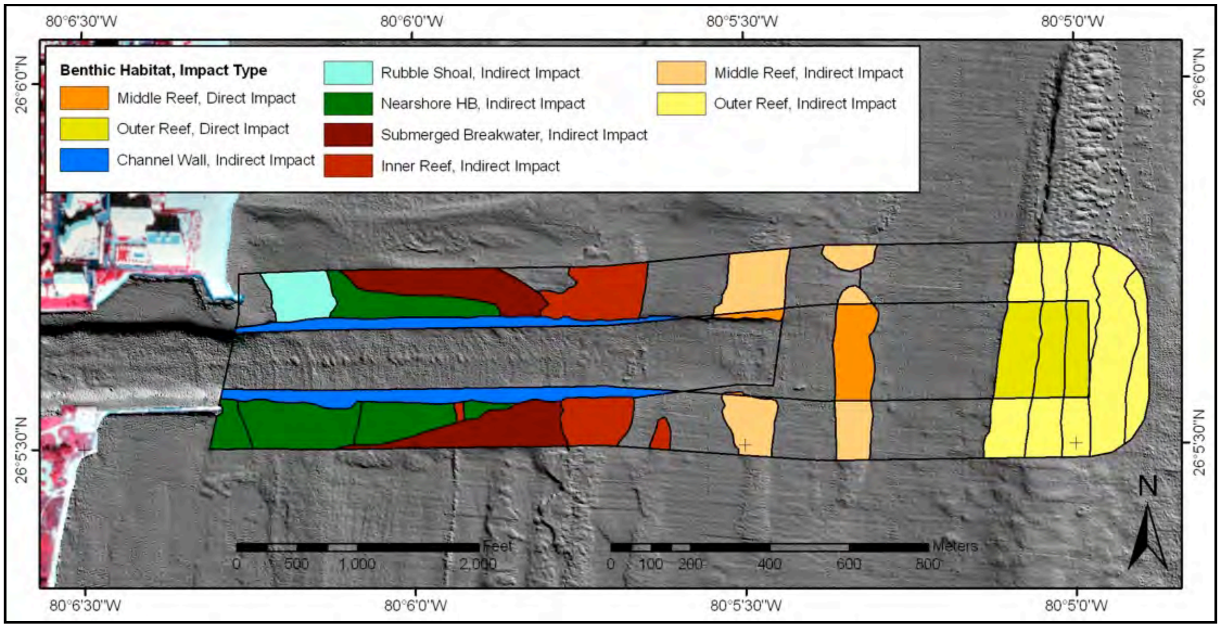

Figure 5. Coral reef habitat impact types within the Port Everglades Expansion Area (from Walker et al. 2008b)

Table 6 summarizes the types, area, and duration of effects to designated critical habitat for elkhorn and staghorn corals. The activities associated with the Port Everglades Expansion project will result in both permanent and temporary effects, as described below. In addition, some of the effects are certain or judged likely to occur (the "confirmed" and "predicted" impacts), while others are contingent upon the ultimate methods used by the contractor (the "potential" impacts). Because we must use a precautionary approach to analyzing effects, we will assume that the maximum potential adverse effects will occur.

Table 6. Summary of Adverse Effects to *Acropora* Designated Critical Habitat from the Port Everglades Expansion Project

Type of Impact	Duration	Area (acres)
Direct removal via explosives and dredging	Confirmed Permanent	15.55
Reef fracturing and sediment/rubble deposition	Confirmed Permanent	6.11
Anchor placement and drag	Potential Permanent	19.31
Sedimentation	Predicted Permanent	1.96
Sedimentation	Predicted Temporary	96.22
Total Impacts		**139.15**

While it may appear that coral reefs and hardbottom habitats are solid rock and extremely structurally stable, the opposite is actually true. Up to 40% of the reef structure may be void spaces because the reef is created by layering dead skeleton, other calcifying organisms, and sediments (Jaap et al. 2006). Therefore, reef and hardbottom (i.e., the essential feature of staghorn critical habitat) is susceptible to damage from physical impacts. The proposed project will permanently adversely affect 21.66 acres of designated critical habitat for staghorn coral. Approximately 15.55 acres of designated critical habitat will be adversely affected through direct removal of the essential feature by explosives and dredging of the middle and outer reef (Figure

12). The underlying reef framework (essential feature) will be permanently destabilized via fracturing and rubble formation, making this area unsuitable and unavailable for coral recruitment and growth. An additional 6.11 acres of critical habitat on the middle and outer reefs located below dredge depth of -57 ft will be impacted due to fracturing of the reef framework and downslope movement of sediments and rubble as a result of dredging. Fracturing the reef framework will permanently destabilize the essential feature rendering it unsuitable and unavailable for coral recruitment and growth. Further, depending on the size and density of the created rubble, it may stay within the impact area indefinitely, also making the area unsuitable for coral recruitment and growth. Similar impacts from ship groundings and explosive use have resulted in significantly lower recruitment rates compared to un-impacted adjacent reef (Fox et al. 2003; Piniak et al. 2010; Rubin et al. 2008). Therefore, we believe that a total of 21.66 acres of designated critical habitat will be permanently adversely affected by the dredging activities.

Based on benthic habitat maps (Walkeret al. 2008b) for the area (including the 150-m indirect impact zone adjacent to the existing channel), the project may potentially permanently impact up to an additional 19.31 acres of critical habitat adjacent to the channel via anchor placement and cable drag (Figure 12, indirect impact areas). The USACE does not anticipate that this impact will occur because the most cost-effective dredging methods will likely avoid these impacts. However, given the potential for these impacts, we are identifying how they may adversely affect critical habitat. Anchor placement and drag may result in the deconsolidation of the hardbottom, rendering it into rubble or smaller fragments. Such impacts can have lasting effects on the physical structure of the site and decrease its abiltiy to support coral recruitment and growth (Rogers and Garrison 2001). Thus, this area would no longer be suitable or available for coral recruitment or growth. So, should a dredging method be selected that results in anchor placement and cable drag, we believe that an additional 19.31 acres of designated critical habitat may be affected.

In addition to the permanent physical impacts from blasting, dredging and/or anchoring identified above, we predict another 98.09 acres of critical habitat in the 150 m areas adjacent to the channel will be impacted by sedimentation caused by dredging. The creation and resuspension of sediments during construction will result in sediment transport and deposition onto the essential feature, rendering it temporarily unsuitable and unavailable for coral recruitment and growth. Sedimentation affects larval settlement and recruitment, and fragment attachment. Sediment accumulation on dead coral skeletons and exposed hard substrate reduces the amount of available substrate suitable for coral larvae settlement and fragment reattachment. Even small increases in sedimentation can significantly reduce coral recruitment and survivorship (Babcock and Smith 2000), and sediments coupled with turf algae further impede recruitment (Birrell et al. 2005). Further supporting the impact sedimentation has on recruitment, coral larvae of some species settle preferentially on vertical surfaces to avoid sediments and cannot successfully establish themselves in shifting sediment (U.S. Army Engineer Research Development Center 2005). Last, survivorship of branching coral fragments is significantly affected by the type of substrate, with increased mortality being linked to the presence of sandy sediments (Lirman 2000). Therefore, if sediments are present and deposited on the area adjacent to the channel, critical habitat may be unavailable for coral larval and fragment recruitment and growth.

Even so, coral reefs are dynamic systems and sediments are often removed from the reef substrate by currents, tides, or storm events, especially those on exposed coasts like the Florida Reef Tract. The residence time of sediments is dependent on several factors including grain size and the hydrodynamics of the system (i.e., higher energy is needed to mobilize larger grained materials). According to the DEIS (USACE 2013), sediment constituents encountered at the Port vary greatly according to location and elevation. The majority of substrate materials within the dredging area include inter-bedded layers of sand and rock. A minority of the material includes silts, clays, and peat/organics. Approximately 80%-90% of the softer excavated rocks are classified as sands with mixed gravel. The harder materials are classified as boulders of varying size. Based on monitoring of nearby beach nourishment projects, it is likely that the impacts of sedimentation are likely to be temporary, with the majority of the area returning to suitable conditions after approximately 18 months (Prekel et al. 2008). Previous monitoring from dredge events at Key West and Port Everglades show no permanent impacts from sedimentation, but some NCRI scientists believe some permanent impacts due to sedimentation may occur from the proposed action. NMFS and USACE agreed meetings held in November 2013 that the majority of the sediment effects are likely to be temporary. To be conservative we will consider a maximum of 2% or 1.96 acres of the area predicted to be impacted by sedimentation will be permanently adversely affected and 96.22 acres of the area predicted to be impacted by sedimentation will only be temporarily adversely affected by dredging. Given that there are no elkhorn or staghorn corals in the area which could use this area for fragment or larvae settlement, we believe that the temporary effects from sedimentation to this 96.22 acres of critical habitat are insignificant.

While there are 133 acres of hard substrate along the bottom and walls of the existing channel, it does not provide the essential feature for *Acropora* settlement and recruitment. As discussed in the final rule designating critical habitat, we determined that existing federally-authorized channels do not provide the essential feature. This is based on the disturbed nature of the substrate within channels and channel walls (i.e., it has been dredged from its natural condition). Further, sediment movement, suspension, and deposition levels are high within existing channels. Hard substrate found within these channels and along their walls are ephemeral in nature and are frequently covered by sand or disturbed by maintenance dredging, thus not meeting the definition of the essential feature. Therefore, the impacts to the hardbottom that occurs in the channel bottom and channel walls are not considered impacts to *Acropora* critical habitat and thus are not part of our critical habitat impact analysis.

Depending on vessel operations and waterway safety, it may be necessary to temporarily or permanently move some or all of the fixed and floating ATONs within the project area (up to 20 total). If ATONs are moved temporarily, all relevant and applicable USCG ATON PDCs (project design criteria) and BMPs (best management practices) will be followed as laid out in NMFS's previous Biological Opinion to the USCG (SER-2011-3196) governing ATON placement and maintenance. This includes the temporary placement of ATONs in areas that are not likely to adversely affect endangered species or habitats. If ATONs are to be moved permanently based on considerations by the U.S. Coast Guard not related to this dredging project, independent consultation with NMFS will take place before the permanent placement of ATON. Generally, fixed ATONs will be removed and replaced with temporary floating ATONs during the dredge project to allow for contractor flexibility and safety of vessels transiting the waterway. The temporary floating ATONs will be placed on the fixed ATONs' prior assigned positions until consultation with NMFS is concluded. ATONs will be temporarily relocated within 30 ft of the existing channel, within the indirect impact zone. Therefore, we believe that effects from the temporary relocation of ATONs will insignificant.

The proposed project includes creation of 5 acres of boulder reef with approximately 12,500 corals relocated from within the dredge footprint. Because the boulder reefs will not be placed on the essential feature of critical habitat, we believe there will be no effect to critical habitat resulting from this activity.

The proposed project also includes enhancement of degraded reef sites with propagation and/or outplanting of additional corals, including 35,000-50,000 colonies of *Acropora cervicornis* at appropriate densities, as mitigation required by the USACE under its authorities to compensate for the impacts to corals and coral reefs. For purposes of this opinion, degraded reef sites are those that are not currently healthy coral dominated reefs due to a previous impact or environmental condition but that could easily be improved through outplanting activities; transplant sites will not include areas with ongoing environmental conditions that would prevent newly outplanted corals from surviving. A comprehensive transplantation and monitoring plan will be developed and approved by NMFS prior to construction to ensure the success of the propagation and outplanting portion of the project. We believe that this portion of the mitigation proposal will have a beneficial effect on designated critical habitat, by accelerating the provision of its intended conservations functions for staghorn coral. The following analysis shows how we determined that the propagation and outplanting component of the project would provide for the conservation of the species.

Facilitating increased incidence of successful sexual and asexual reproduction is the key objective to the conservation[10] of staghorn coral identified for its designated critical habitat (73 FR 72224, November 26, 2008), based on the species' life history characteristics, population declines, and extremely low recruitment. Therefore, the critical habitat designation identifies the essential feature within the areas occupied by the species that need protection to support that goal. Corals are sessile and depend upon external fertilization in order to produce larvae. Fertilization success is reduced as adult density declines (known as the Allee effect) (Levitan 1991). Since *Acropora* is not able to self-fertilize it requires a certain density (discussed in further detail below) of adult colonies to promote sexual reproduction (*Acropora* Biological Review Team 2005).

Another activity that supports the goal of increased incidence of successful sexual and asexual reproduction is artificial propagation of the species. The Recovery Outline for Elkhorn and Staghorn Coral (NMFS 2013) identifies the following key action necessary to promote conservation:

> *Develop and implement appropriate strategies for population enhancement, through restocking and active management, in the short to medium term, to increase the likelihood of successful sexual reproduction and to increase wild populations.*

[10] Under the ESA, conservation is equated with recovery of a species (i.e., the species no longer needs the protection of the ESA).

Numerous nurseries for staghorn coral have been established to support this recovery activity in the past 15 years with the expressed purpose of enhancing wild populations with sufficient densities of the species to promote natural sexual reproduction (Johnson et al. 2011). To date, hundreds of thousands of staghorn corals have been propagated and outplanted throughout the species' range, with high survival rates (i.e., 75%-90%; T. Moore, NOAA Restoration Center pers. comm. to J. Moore, NMFS PRD, January 22, 2014). Therefore, we are highly confident that propagation and outplanting of staghorn corals support the intended goal.

One of the objectives identified in the Recovery Outline is to ensure the population viability of each species. The NMFS *Acropora* recovery team, working on a draft recovery plan for elkhorn and staghorn corals, has determined that population viability for staghorn coral requires achieving a density of one colony ($\geq$ 0.5 m diameter in size) per square meter, throughout approximately 5% of consolidated reef habitat in 5-20 m water depth throughout the species' range (A. Moulding, NMFS Recovery Team liason pers. comm. To K. Logan, NMFS PRD, February 2014). Based on estimates of the proportion of habitat historically occupied by staghorn thickets, the recovery team has determined that this is the density of adult staghorn coral colonies necessary to facilitate sustained sexual reproduction. We assume that the maximum conservation potential of critical habitat can be calculated by applying this metric of a recovered population. Therefore, we applied this criterion to the area of critical habitat predicted to be permanently adversely affected by the proposed action, to calculate the number of colonies of certain size and density the area would have needed to support, to fulfill the population viability requirements identified by the recovery team. First we determined the proportion of the area that will be permanently adversely affected that would satisfy the habitat requirement, by calculating the acreage representing 5% of the permanently adversely affected area. This results in an area of 4,382.7 m^2 (5% of 23.62 acres = 1.181 acres = 4,779.34 m^2). To determine the size and density requirements, we considered that a colony 0.5 m in diameter will occupy 0.2 m^2 if we assume the colony is roughly circular in shape (area of a circle = $3.14 \times r^2 = 3.14 \times (0.25)^2 = 0.2\ m^2$). Consequently, 0.2 m^2 coral occupancy per square meter of hardbottom is necessary to achieve the size and density goal identified by the recovery team, and to achieve full functionality of critical habitat. The staghorn colonies required to be outplanted by the blended mitigation agreement will be approximately 0.2 m (20 cm) in diameter. Therefore, again assuming the colonies are roughly circular in shape and applying the equation for the area of a circle, the area of an outplanted colony will be 0.03 m^2 ($3.14 \times (0.1)^2 = 0.03\ m^2$). Consequently, approximately 7 colonies of staghorn coral per square meter of hardbottom would be required to provide the full conservation benefit of the critical habitat which will be permanently lost due to the project ($0.2\ m^2/0.03\ m^2 = 6.67$ colonies/m^2). This is consistent with data presented by Vargas-Angel et al. (2003), who have determined that the highest average cover in surveyed staghorn thickets was 25.9%, and the highest average density was 3.3 colonies per m^2 (average colony size 40.8 cm). Multiplying the habitat requirement calculated above (4,779.34 m^2) by the number of colonies needed per square meter (6.67 colonies) results in a total of 31,878 staghorn colonies. Further calculations regarding recruitment, mortality, and growth rates support this conclusion (see Appendix C).

6.4 Effects of the Action on Staghorn Coral

The blended mitigation plan includes using coral nurseries to grow and subsequently outplant between 35,000 and 50,000 colonies of staghorn coral at appropriate densities. Corals may be collected from existing nurseries or from "corals of opportunity" (i.e., unattached wild colonies or those proposed to be impacted by an authorized activity that can be rescued and relocated). Collecting coral fragments involves directed take (via collection) of *A. cervicornis*. However, the protective regulations pursuant to ESA Section 4(d) for staghorn provides for certain exceptions to the ESA Section 9 prohibitions for scientific research and species enhancement, and restoration carried out by authorized personnel (73 FR 64264; October 29, 2008). Thus, the take that may result from this project's propagation and outplanting of staghorn corals is currently not prohibited, as long as the actions are carried out pursuant to: (1) the exceptions in the 4(d) rule; and (2) the Biological Opinion on the issuance of the rule. Because all activities related to coral propagation (i.e., wild collection, nursery establishment and operation, and outplanting) in Broward County require a State of Florida Special Activity License (SAL), the USACE will be required to hold a valid permit (SAL) and they will be in compliance with the 4(d) rule. However, NMFS has proposed to reclassify staghorn coral from threatened to endangered and proposed to list 6 additional coral species that occur within the action area. Should that proposal become final (decision due June 2014), the aforementioned 4(d) rule for staghorn corals will be void because there are no exceptions to the take prohibitions allowed for endangered corals. Therefore, the take that will result from the propagation activities will need authorization. If the proposed reclassification is finalized in June 2014, the USACE will need to contact NMFS to determine the mechanism for authorizing the take of corals necessary to implement this action as proposed.

While the take of staghorn coral is not currently prohibited, we must still include the take that will result from coral nurseries in the evaluation of whether the proposed action will jeopardize the continued existence of the species.

Active coral propagation has been identified as a priority for staghorn coral by NMFS and by the *Acropora* Recovery Team. Over the course of the mitigation portion of the project, it is likely that fragments will be taken from fewer than 250 wild healthy colonies and brought into nurseries. The rest will be sourced from "corals of opportunity." Typically, collection of donor coral fragments is only necessary during the first year of a nursery. No additional coral collection is required after the first year of establishing a nursery since the nurseries produce enough coral tissue for both expansion and outplanting. Typically, approximately 20% of the corals in the nursery are designated to serve as broodstock while the remaining 80% will be outplanted. The broodstock corals are divided into multiple segments/fragments, which are maintained and grow in the nursery until they are ready to be outplanted.

NMFS believes that the collection of small fragments from wild *A. cervicornis* colonies will result in temporary effects on coral colonies. The collection of branch tip fragments from single staghorn coral colonies will result in a small reduction of coral colony biomass; however, this effect is expected to be temporary with recovery through tissue replacement and/or coral colony growth. *Acropora cervicornis*' dominant mode of reproduction is through asexual fragmentation. In the congener *Acropora palmata*, lesions at the point of fragment detachment have been shown to begin regeneration within 2 weeks of fragmentation (Lirman 2000), with regeneration rates being positively correlated with decreasing size of lesion and proximity to growing tip. The size of the lesion created in this project will be a function of the diameter of the branch being clipped. The diameter of staghorn coral branches ranges from 0.25 to 1.5 cm. Lirman (2000) showed that a 3-cm^2 lesion regenerated completely within 100 days. Given that the rate of recovery is an exponential decay, it is expected that lesions 0.25 to 1.5 cm in diameter (less than 2.25 cm^2) will recover much faster than in Lirman's experiment.

Furthermore, the proposed collection of fragments from *A. cervicornis* colonies will occur at the outermost portion of the branch tip of the coral colony. Soong and Lang (1992) observed that, in *A. cervicornis*, large polyps and basal tissues located 1.0 to 4.5 cm from the colony base were infertile, and larger eggs were located in the mid-region of colony branches. Gonads located within 2 to 6 cm of the colony's branch tips always had smaller eggs than those in the mid-region (Soong and Lang 1992). Larger colonies (as measured by surface area of the live colony) have higher fertility rates (Soong and Lang 1992). Thus, the effect of this activity on coral colony reproduction is insignificant. Given that the collected tissue samples are small in size (~20 cm) relative to coral colony size, that the effects of collecting such fragments are temporary, that fragmentation is a natural reproductive mode, and that these fragments will be collected from the outermost portion of the coral branch tip where smaller eggs are found, it is not likely that survival or reproductive output of staghorn coral colonies will be measurably reduced by the collection of staghorn fragments for nursery propagation.

The blended mitigation plan estimates that between 35,000 and 50,000 colonies of staghorn coral will be produced and outplanted to degraded reef sites, in the sizes and densities discussed above as needed to facilitate sustained, successful sexual reproduction. These colonies will supplement the wild populations within Broward County. Successful sexual reproduction is a goal of the recovery outline and identified as the key conservation goal of the critical habitat designation for staghorn (and elkhorn) corals. The purpose of outplanting staghorn coral into the wild is to enhance the wild population and provide additional potential for successful sexual reproduction. Outplanting will achieve the proper density and provide a source of varied genetic material which will increase the likelihood of sexual reproduction. Therefore, the survival and reproductive potential of staghorn coral will be enhanced by this action.

6.5 Effects of the Action on Proposed Coral Species

The analyses in this section are based upon the best available biological data on the proposed coral species and the effects of the proposed action. Data pertaining to effects from the proposed action relative to interactions with proposed species are limited. In such circumstances, we are often forced to make assumptions to overcome the limits in our knowledge. Frequently, different analytical approaches may be applied to the same data sets. In those cases, in keeping with the direction from the U.S. Congress to resolve uncertainty by providing the "benefit of the doubt" to threatened and endangered species [House of Representatives Conference Report No. 697, 96th Congress, Second Session, 12 (1979)], we will generally select the value yielding the most conservative outcome (i.e., the value which would lead to conclusions of higher, rather than lower, risk to endangered or threatened species).

We believe the proposed project will adversely affect 6 coral species that are proposed to be listed under the ESA (elliptical star coral, Lamarck's sheet coral, rough cactus coral, mountainous star coral, knobby star coral, and lobed star coral). Table 7 summarizes our estimates of the number of colonies of each proposed coral species that occur in the direct and indirect impacts areas. These estimates were calculated by applying the average species densities based on survey data provided by Dial Cordy, Inc. to each of the impact areas. In order to estimate the numbers of *O. annularis, O.fanksi,* and *O. faveolata* (because the Dial Cordy survey only identified the *Orbicella* complex) we applied the species densities from the study completed by Gillam and Walker (2011) to the total number of *Orbicella* complex identified in the Dial Cordy, Inc. survey area. The Dial Cordy, Inc. survey was only conducted in the middle- and out-reef areas. No surveys have been conducted within the channel bottom and channel walls. Therefore, to be conservative we are applying the densities of the proposed corals from the middle and outer reefs to the channel and channel wall hardbottom. However, it is unlikely that the proposed corals occur at the same densities as on the reef itself. Due to the shipping activity in the channel, there is likely much poorer water quality conditions within the channel as compared to the reef. Therefore, we assume the coral densities are likely much lower. Further, the channel has been dredged within the last 30 years. Given the relatively slow growth rates of the proposed corals, it is likely that the colonies that do exist within the channel and channel walls are smaller sizes than those on the reef. Thus, we anticipate that the estimates we provide for mortality of proposed corals within the channel and channel walls are likely an overstimate; however, it is the best available information and provides a conservative assessment of impacts to the species.

Table 7. Estimated Proposed Coral Colonies Within the Impact Area

Proposed Coral Species	Mortality (Middle and Outer Reef <57ft 15.55 ac)	Relocation Survival (Middle and Outer Reef 15.55 ac)	Relocation Mortality (Middle and Outer Reef 15.55 ac)	Mortality (Middle and Outer Reef >57ft 6.11 ac)	Mortality (Channel Bottom and Walls 133 ac)	Mortality (Indirect Impact Area 1.96 acres)	Mortality Total
Lamarck's sheet	0	35	6	16	352	5	379
Elliptical star	1522	105	19	646	14,071	207	16465
Lobed star	1121	773	657	792	17,238	254	20062
Mountainous star	36	25	21	29	24	517	627
Knobby star	36	25	21	29	24	517	627
Rough cactus	82	35	6	48	1,055	16	1207

We assume that all the proposed corals that occur in impact areas other than the middle and outer reef shallower than 57 ft will be killed as a result of the dredging operations. The USACE has proposed to relocate all proposed corals greater than or equal to 10 cm longest linear dimension from the middle and outer reef impact areas shallower than 57 ft.

Even though the relocation of proposed coral colonies involve directed take (collection), the USACE has proposed the relocation because the effect to the species is significantly reduced as compared to the level of almost certain lethal take of the proposed coral that would occur through direct removal via dredging, anchor placement, and cable drag. Relocations will result in: (1) a high likelihood of continued survival of the coral transplants, (2) the survival of the unique genetic material of the transplanted colonies, and (3) the potential for use of the material in future restoration activities. The Consultation Handbook (USFWS and NMFS 1998) expressly authorizes such directed take as an RPM (see page 4-53). Therefore, NMFS will evaluate the expected level of take through relocation so that these levels can be included in the evaluation of whether the proposed action will jeopardize the continued existence of the species.

Coral transplantation can successfully relocate colonies that would likely suffer injury or morality if not moved. Thornton et al. (2000) documented a 13% mortality rate for transplanted scleractinian corals in southeast Florida. The high rate of survival is attributed to the methods used and life history of corals. Lindahl (2003) showed that skilled handling does not significantly affect coral fragments or, by extension, coral colonies. Many different species of coral have shown high survival after transplantation, provided that colonies are handled with skill, are reattached properly, and the environmental conditions at the reattachment site are conducive to their growth (Maragos 1974, Birkeland et al. 1979, Harriott and Fisk 1988, Hudson and Diaz 1988, Guzman 1991, Kaly 1995, Becker and Mueller 1999, Tomlinson and Pratt 1999, Hudson 2000, Lindahl 2003, NCRI 2004).

The USACE and NMFS agree that all of the colonies of elliptical star, mountainous star, knobby star, lobed star, rough cactus, and Lamarck's sheet coral could be lethally taken during dredging if not relocated. Therefore, the USACE is proposing to relocate all colonies over 10 cm. We believe coral transplantation will be highly successful and relocating these corals outside the project area is an appropriate alternative to the take that would otherwise occur. The corals will be transplanted to the newly created artificial reef nearby the proposed project. Corals will be transplanted using the appropriate transplantation protocols (see Appendix B) by properly trained personnel. Corals will be placed on the artificial reef in area appropriate densities and grouped by species. Because suitable transplantation habitat is nearby and proper handling techniques are available and will be required, we have confidence that transplantation survival rates similar to those noted elsewhere will be likely in this case. We believe that a 15% coral morality rate of these corals being transplanted from their natural environment to areas nearby is a reasonable estimate; therefore, we anticipate an 85% survival rate of transplanted colonies.

The mitigation plan also includes the propagation and outplanting of corals to compensate for the impacts to corals and coral reef habitats. This portion of the mitigation plan is not finalized; therefore, it is unknown if any of the proposed species will be affected by this activity. None of the proposed coral species is currently in active propagation in any of the existing coral nurseries in Broward County. As described in Section 6.4, coral propagation and outplanting is beneficial to corals despite the initial take required to begin the nursery operations. Therefore, should any of the proposed species be propagated as part of the mitigation plan, the effects to them would also be beneficial.

In summary, we estimate that a maximum of 379 colonies of Lamarck's sheet coral, 16,465 colonies of elliptical coral, 20,062 colonies of lobed coral, 627 colonies of mountainous coral, 627colonies of knobby star coral, and 1207 colonies of rough cactus coral will be lethally taken during dredging activities. We also estimate that a maximum of 35 colonies of Lamarck's sheet coral, 105 colonies of elliptical coral, 773 colonies of lobed coral, 25 colonies of mountainous coral, 25 colonies of knobby star coral, and 35 colonies of rough cactus coral will be relocated and survive.

7 Cumulative Effects

Cumulative effects include the effects of *future* state, tribal, or local private actions – i.e., that are not already in the baseline -- that are reasonably certain to occur in the action area considered in this opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to Section 7 of the ESA (50

CFR 402.14). Actions that are reasonably certain to occur would include actions that have some demonstrable commitment to their implementation, such as funding, contracts, agreements or plans.

NMFS is aware of several future projects that may contribute to cumulative effects. Broward County is planning to begin construction on a mangrove enhancement project directly adjacent to the proposed Port expansion project. The County and Port also plan to expand the turning notch under a separate project. These activities will impact mangroves and may also impact Johnson's seagrass and sea turtles depending on the final construction methodology.

Within the action area, major future changes are not anticipated in addition to the ongoing human activities described in the environmental baseline. The present human uses of the action area, such as commercial shipping, are expected to continue, though some may occur at increased levels, frequency or intensity in the near future.

8 Jeopardy Analysis

The analyses conducted in the previous sections of this opinion provide the basis on which we determine whether the proposed action would be likely to jeopardize the continued existence of green and loggerhead sea turtles, Johnson's seagrass, staghorn coral, and corals proposed for ESA listing. In Section 6, we outlined how the proposed action would affect these species at the individual level and the magnitude of those effects based on the best available data. Next, we assess each of these species' response to the effects of the proposed action, in terms of overall population effects, and whether those effects will jeopardize their continued existence in the context of the status of the species (Section 4), the environmental baseline (Section 5), and the cumulative effects (Section 7).

It is the responsibility of the action agency to "insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered species or threatened species…" (ESA Section 7(a)(2)). Action agencies must consult with and seek assistance from the NMFS to meet this responsibility. NMFS must ultimately determine in a Biological Opinion whether the action jeopardizes listed species. To *jeopardize the continued existence of* is defined as "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). The following jeopardy analysis first considers the effects of the action to determine if we would reasonably expect the action to result in reductions in reproduction, numbers, or distribution of loggerhead and green sea turtles, Johnson's seagrass, staghorn coral, or proposed coral species. The analysis next considers whether any such reduction would in turn result in an appreciable reduction in the likelihood of survival of these species in the wild, and the likelihood of recovery of these species in the wild.

8.1 Green Turtles

The potential lethal take of up to 2 green sea turtles (1 observed and 1 unobserved) by hopper dredge is a reduction in numbers. These lethal takes would also result in a potential reduction in future reproduction, assuming some individuals would be females and would have survived otherwise to reproduce. All life stages are important to the survival and recovery of sea turtles; however, it is important to note that individuals of one life stage are not equivalent to those of other life stages. For example, the take of male juveniles may affect survivorship and recruitment rates into the reproductive population in any given year, and yet not significantly reduce the reproductive potential of the population. A very low percent of hatchlings is typically expected to survive to reproductive age. The death of mature, breeding females can have an immediate effect on the reproductive rate of the species. Sublethal effects on adult females may also reduce reproduction by hindering foraging success, as sufficient energy reserves are probably necessary for producing multiple clutches of eggs in a breeding year. Different age classes may experience varying rates of mortality and resilience. Further, an adult green sea turtle can lay 1-7 clutches (usually 2-3) of eggs every 2-4 years, with 110-115 eggs/nest of which a small percentage is expected to survive to sexual maturity. Green sea turtles are highly migratory, and individuals from all Atlantic nesting populations may range throughout the Gulf of Mexico, Atlantic Ocean, and Caribbean Sea. Because all the potential interactions are expected to occur at random throughout the proposed action area and sea turtles generally have large ranges in which they disperse, the distribution of green sea turtles in the action area is expected to be unaffected.

To be conservative, we assume that the green sea turtles that will be taken will be reproductive females, with a higher potential impact on the species relative to take of other stages. If the take is of a reproducing female, it is likely that such a turtle is part of the Florida population (female returning to nesting beach).

This species is currently showing a very large increasing nesting trend in Florida, with nesting numbers already approaching or exceeding those required by the recovery plan for the species. Therefore, we believe that the reduction in numbers and reproduction as a result of the lethal take is not expected to appreciably reduce the likelihood of survival of green sea turtles in the wild.

We also considered the recovery objectives in the recovery plan prepared for the U.S. populations of green sea turtles that may be affected by the predicted reduction in numbers and reproduction. The recovery plan for green sea turtles (NMFS and USFWS 1991) lists the following relevant recovery objectives relevant to the effects of the proposed action:

> <u>• The level of nesting in Florida has increased to an average of 5,000 nests per year for at least 6 years. Nesting data must be based on standardized surveys</u>. Between 2001 and 2006, an average of 5,039 green turtle nests were laid annually in Florida, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). That average increased to 7,436 nests per year for the 6-year period of 2004-2009. Data from the index nesting beach program in Florida support the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, but that is thought to be part of the normal biennial nesting cycle for green turtles (FWC Index Nesting Beach Survey Database). An additional drop to just below 3,000 nests was seen on the index nesting beaches in 2009, but the occasional break from the normal biennial pattern is not without precedent, as there were 2 consecutive years of increase from 2003-2005 (FWC Index Nesting Beach Survey Database). State nesting data for 2011 show an

increase in green turtle nests to 10,701, the highest number of nests since 1988 (FWRI Web site: http://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/).

• A reduction in stage class mortality is reflected in higher counts of individuals on foraging grounds. Currently, there are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas (where they come to forage) of the southeastern United States. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant (they have averaged 215 green sea turtle captures per year since 1977) in St. Lucie County, Florida, show that the annual number of immature green sea turtles captured has increased significantly in the past 26 years (FPL 2002). Ehrhart et al. (2007) has also documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area.

The lethal take of 2 turtles will result in a reduction in numbers and reproduction, but will not have any detectable influence on the population and nesting trends noted above. The loss of 2 individuals will not have an appreciable impact on total recruitment of new sea turtles to the population given the extent of the impact versus the very rapid population increases occurring over the past decade. Thus, the proposed action will not interfere with achieving the recovery objectives above and will not result in an appreciable reduction in the likelihood of green sea turtles' recovery in the wild.

8.2 Loggerhead Turtles (NWA DPS)

The potential lethal take of up to 2 loggerhead sea turtles (1 observed and 1 unobserved) by hopper dredge is a reduction in numbers. These lethal takes would also result in a reduction in reproduction as a result of lost reproductive potential, as some of these individuals would be females who would have survived other threats and reproduced in the future, thus eliminating each female individual's contribution to future generations. All life stages are important to the survival and recovery of sea turtles; however, it is important to note that individuals of one life stage are not equivalent to those of other life stages. For example, the take of male juveniles may affect survivorship and recruitment rates into the reproductive population in any given year, and yet not significantly reduce the reproductive potential of the population. A very low percent of hatchlings is typically expected to survive to reproductive age. The death of mature, breeding females can have an immediate effect on the reproductive rate of the species. Sublethal effects on adult females may also reduce reproduction by hindering foraging success, as sufficient energy reserves are probably necessary for producing multiple clutches of eggs in a breeding year. Different age classes may experience varying rates of mortality and resilience. Further, an adult female loggerhead sea turtle can lay 3-4 clutches of eggs every 2-4 years, with 100 to 130 eggs per clutch. The annual loss of adult female sea turtles, on average, could preclude the production of thousands of eggs and hatchlings of which a small percentage would be expected to survive to sexual maturity. A reduction in the distribution of loggerhead sea turtles is not expected from lethal takes during the proposed action. Because all the potential interactions are expected to occur at random throughout the proposed action area and sea turtles generally have large ranges in which they disperse, the distribution of loggerhead sea turtles in the action area is expected to be unaffected.

Whether or not the reductions in loggerhead sea turtle numbers and reproduction attributed to the proposed action would appreciably reduce the likelihood of survival for loggerheads depends on what effect these reductions in numbers and reproduction would have on overall population sizes and trends, i.e., whether the estimated reductions, when viewed within the context of the environmental baseline and status of the species, are of such an extent that adverse effects on population dynamics are appreciable. In Section 3.2.2, we reviewed the status of the species in terms of nesting and female population trends and several recent assessments based on population modeling [i.e., (Conant et al. 2009; NMFS-SEFSC 2009d)]. Below we synthesize what that information means in general terms and also in the more specific context of the proposed action.

Loggerhead sea turtles are a slow growing, late-maturing species. Because of their longevity, loggerhead sea turtles require high survival rates throughout their life to maintain a population. In other words, late-maturing species cannot tolerate much anthropogenic mortality without going into decline. Conant et al. (2009) concluded loggerhead natural growth rates are small; natural survival needs to be high; and even low to moderate mortality can drive the population into decline. Because recruitment to the adult population is slow, population modeling studies suggest even small increased mortality rates in adults and subadults could substantially impact population numbers and viability (Chaloupka and Musick 1997; Crouse et al. 1987; Crowder et al. 1994; Heppell et al. 1995).

The best available information indicates that the NWA loggerhead DPS is still large, but is possibly experiencing more mortality than it can withstand. All of the results of population models in both NMFS SEFSC (2009d) and Conant et al. (2009) indicated western North Atlantic loggerheads were likely to continue to decline in the future unless action was taken to reduce anthropogenic mortality. With the inclusion of newer nesting data beyond the 2007 data used in those analyses, the status of loggerhead nesting is beginning to show improvement. As previously described in the Status of the Species section, in 2008 nesting numbers were high, but not enough to change the negative trend line. Nesting dipped again in 2009, but rose substantially in 2010. With the addition of data through 2010, the nesting trend for the NWA DPS of loggerheads is only slightly negative and not statistically different from zero (no trend) (NMFS and USFWS 2010). Additionally, although the best fit trend line is slightly negative, the range from the statistical analysis of the nesting trend includes both negative and positive growth (NMFS and USFWS 2010). The 2011 nesting was on par with 2010, providing further evidence that the nesting trend may have stabilized and the 2012 index nesting number was the largest since 2000.

To be conservative, we assume that the loggerhead sea turtles that will be taken will be reproductive females, with a higher potential impact on the species relative to take of other stages.

NMFS SEFSC (2009d) estimated the minimum adult female population size for the western North Atlantic in the 2004-2008 time frame to likely be between 20,000 to 40,000 (median 30,050) individuals, with a low likelihood of being as many as 70,000 individuals. Estimates were based on the following equation: Adult females = (nests/(nests per female)) x remigration interval. The estimate of western North Atlantic adult loggerhead female was considered conservative for several reasons. The number of nests used for the western North Atlantic was based primarily on U.S. nesting beaches. Thus, the results are a slight underestimate of total nests because of the inability to collect complete nest counts for many non-U.S. nesting beaches. In estimating the current population size for adult nesting female loggerhead sea turtles, NMFS SEFSC (2009d) simplified the number of assumptions and reduced uncertainty by using the minimum total annual nest count over the relevant 5-year period (2004-2008) (i.e., 48,252 nests). This was a particularly conservative assumption considering how the number of nests and nesting females can vary widely from year to year (cf., 2008's nest count of 69,668 nests, which would have increased the adult female estimate proportionately, to between 30,000 and 60,000). In addition, minimal assumptions were made about the distribution of remigration intervals and nests per female parameters, which are fairly robust and well known parameters. Florida's long-term loggerhead nesting data (1989-2012) has shown three distinct trends. Following a 23% increase between 1989 and 1998, nest counts declined sharply for over a decade. During the period between the high-count nesting season in 1998 and the most recent (2012) nesting season, researchers found no demonstrable trend, indicating a reversal of the post-1998 decline. The overall change in counts from 1989 to 2012 is positive. Nest counts in 2012, corrected for subtle variation in survey effort, were slightly below the high nest count recorded in 1998.

Based on the total numbers of adult females estimated by NMFS SEFSC for the western North Atlantic population of loggerhead sea turtles, the anticipated lethal take of 2 loggerheads – in the extremely unlikely worst case that both are female and adult –resulting from the proposed action would represent the removal of approximately 0.006% ([2/30,000] x 100) of the estimated adult loggerhead female population. These removals are very small and contribute only minimally to the overall mortality on the population. Further, these percentages are likely an overestimation of the impact of the anticipated lethal take resulting from the proposed project on loggerhead sea turtles for the following reason. These percentages represent impacts to adult female loggerhead sea turtles only, and not to the population as a whole. Because this estimated contribution to mortality is a tiny part of our range of uncertainty across what total mortality might be for loggerhead sea turtles, we believe that the small effect posed by the lethal take resulting from the proposed project will not result in a detectable or appreciable reduction in the species' likelihood of survival in the wild.

We also considered the recovery objectives in the recovery plan prepared for the U.S. populations of loggerhead sea turtles that may be affected by the predicted reduction in numbers and reproduction. The Services' recovery plan for the Northwest Atlantic population of the loggerhead turtle (NMFS and USFWS 2009), which is in essence the same population of turtles as comprise the NWA DPS, provides explanation of the goals and vision for recovery for this population. The objectives of the recovery plan most pertinent to the threats posed by dredging associated activities are numbers 11 and 13:

11. Minimize trophic changes from fishery harvest and habitat alteration…
13. Minimize vessel strike mortality.

As discussed above, the proposed action will remove several acres of foraging habitat for sea turtles; however, the project area is surrounded by abundant seagrass meadows and the channel slopes will be recolonized by epifauna and flora once the dredging has concluded. Therefore, there will be insignificant effects from permanent loss of habitat that may have been used for foraging by sea turtles. Thus, the action will not interfere with achieving Objective 11. The take predicted from the action is entrainment of turtles by hopper dredges and thus does not constitute vessel strike mortality as envisioned in the recovery plan. Further, the proposed action is expected to reduce the level of vessel traffic using the inlet and harbor (fewer, larger vessels are anticipated). Further, since some of the larger vessels are already coming in at high tide with the narrow channels, there is a greater chance of turtles being struck since turtles don't have adequate room to move away from an oncoming ship. The widening and deepening should help to provide more room for turtles to avoid ships. Thus, the proposed action will not interfere with achieving Objective 13.

The recovery plan anticipates that, with implementation of the plan, the western North Atlantic population will recover within 50 to 150 years, but notes that reaching recovery in only 50 years would require a rapid reversal of the declining trends of the Northern, Peninsular Florida, and Northern Gulf of Mexico Recovery Units. The potential lethal take of 2 loggerheads during the project will result in reduction in numbers when take occurs and possibly by lost future reproduction, but given the magnitude of these trends and likely large absolute population size, it is unlikely to have any detectable influence on the population objectives and trends noted above. Loggerhead nest counts on Florida's index beaches have declined from a peak of nearly 60,000 in 1998. However, 2011 counts were close to the average of the previous 5 years. Although this may be the beginning of a stabilizing trend, additional good nesting years will be required to reverse the preceding decline (FWRI Web site: http://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/).

Thus, the proposed action will not interfere with achieving the recovery objectives and will not result in an appreciable reduction in the likelihood of loggerhead sea turtles' recovery in the wild.

8.3 Johnson's seagrass

The estimated loss of up to 4.67 acres of Johnson's seagrass due to the proposed action is a conservative, reasonable worst-case scenario. The actual amount is likely much lower, but to be conservative, we assumed that all of the mixed beds contained 50% coverage of Johnson's seagrass. The loss of 4.67 acres of Johnson's seagrass is a reduction in numbers of the species. However, in terms of adverse effects on a larger, population scale, the Johnson's Seagrass Recovery Team determined that effects of dredging and filling activities are generally local and small -scale in nature and are not considered threats to the survival and recovery of the species. These activities will not individually or cumulatively result in the long -term, large -scale mortality of Johnson's seagrass, particularly in light of its "pulsating patches" life history strategy, discussed above. Thus, although up to 4.67 acres of Johnson's seagrass will be lost in the immediate action area, the project will not result in any adverse effects on a larger, population scale.

Reproduction will be reduced by the up to 4.67-acre reduction in Johnson's seagrass numbers, but NMFS considers that this reproductive loss does not appreciably reduce the likelihood of survival of Johnson's seagrass in the wild. Johnson's seagrass will continue to reproduce and spread because the proposed impacts are localized and will not affect any Johnson's seagrass outside of the dredge footprint. Johnsons's seagrass exists in the Dania Cutoff Canal, south of the action area, and will not be impacted.

The proposed action will not result in a reduction of Johnson's seagrass distribution or fragmentation of the range since we expect Johnson's seagrass will persist outside of the action area (in the Dania Cutoff Canal to the south) and will continue to be capable of spreading via asexual fragmentation. Therefore, the reproductive potential of the species in this portion of its range will persist.

Recovery for Johnson's seagrass, as described in the recovery plan, will be achieved when the following recovery objectives are met: (1) the species' present geographic range remains stable for at least 10 years, or increases; (2) self-sustaining populations are present throughout the range at distances less than or equal to the maximum dispersal distance to allow for stable vegetative recruitment and genetic diversity; and (3) populations and supporting habitat in its geographic range have long-term protection (through regulatory action or purchase acquisition).
NMFS believes that the proposed action will not appreciably reduce the likelihood of recovery of Johnson's seagrass in the wild. NMFS' s 2007 5-year review of the status of the species concluded that the first recovery objective has been achieved. In fact, the range has increased slightly northward. The proposed action will not impact the status of this objective. Self-sustaining populations are present throughout the range of the species. The species' overall reproductive capacity will be only minimally reduced by the reduction in Johnson's seagrass numbers and reproduction resulting from the action. The proposed dredging will not lead to separation of self -sustaining Johnson's seagrass patches to an extent that might lead to adverse effects to one or more patches of the species. Similarly, the availability of suitable habitat in which the species can spread/flow in the future will not be adversely affected by the proposed action. While additional individual impacts may continue to occur, over the last decade the species has not demonstrated any declining trends. The proposed action will not reduce or destabilize the present range of Johnson's seagrass. Therefore, the project will not appreciably reduce the likelihood of recovery of Johnson's seagrass in the wild.

8.4 Staghorn and Proposed Corals

In the following analysis, we evaluate the effects of the lethal take and nonlethal relocation of proposed corals from the Port Everglades Channel and the nonlethal collection of staghorn coral fragments for propagation and outplanting. Over the course of the Port Expansion activities and the 7-year mitigation project, we do not expect the proposed action to have any measurable impact on the reproduction, numbers, or distribution of the species.

As discussed in Section 6 (Effects of the Action), the expansion of Port Everglades is likely to adversely affect a maximum of 379 colonies of Lamarck's sheet coral, 16,465 colonies of elliptical coral, 20,062 colonies of lobed coral, 627 colonies of mountainous coral, 627colonies of knobby star coral, and 1207 colonies of rough cactus coral, by lethal take during dredging activities. However, the majority of the lethal take results from estimating the number of colonies that occur within the channel. We also estimate that a maximum of 35 colonies of Lamarck's sheet coral, 105 colonies of elliptical coral, 773 colonies of lobed coral, 25 colonies of mountainous coral, 25 colonies of knobby star coral, and 35 colonies of rough cactus coral will be relocated and survive.

The proposed action may also collect up to 250 fragments from wild colonies of staghorn coral and collect approximately 2,500 staghorn coral fragments of opportunity to support the propagation and outplanting portion of the mitigation plan.

We must now determine if the action would reasonably be expected to appreciably reduce, either directly or indirectly, the likelihood of staghorn coral or any of the proposed coral's survival and recovery in the wild.

Proposed Corals

Since the final listing has not yet been published, a recovery plan is not available for any of the proposed species. However, we can assess the effects of the proposed action on each of the proposed coral's populations in the context of our knowledge of the statuses of the species and their environmental baselines.

Lamarck's Sheet Coral.

The proposed action will not affect the species' current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 379 colonies would result in a reduction in Lamarck's sheet coral distribution in the immediate action area. However, the species is found throughout the wider Caribbean region. In Florida, Lamarck's sheet coral is found from Palm Beach County through Monroe County. The action area for this project is located in the middle of this range. The proposed action will not result in a reduction of Lamarck's sheet coral distribution or fragmentation of the range since we expect Lamarck's sheet coral will persist within the action area due to relocation of colonies (from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in Lamarck's sheet coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction. We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species, such as substrate availability, community structure, grazing pressure, fecundity, mode, and timing of reproduction. The anticipated loss of 379 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in Lamarck's sheet coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the Lamarck's sheet coral colonies occur in the smaller size classes and no corals were observed larger than 40-cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5-1 cm/yr) we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of Lamarck's sheet corals in the wild.

An estimated maximum of 379 colonies of Lamarck's sheet coral will be lethally taken during dredging activities. While we do not have exact population estimates for this species, a high number of colonies are believed to be still in existence through the species' range. *Agaricia lamarcki* has been reported to be common (Veron, 2000). A 2011 survey conducted by Nova Southeastern University just south of Port Everglades has identified 912 colonies of Lamarck's sheet coral over just 735 acres. On reefs at 30–40 m depths in the Netherlands Antilles, *Agaricia lamarcki* has increased (Bak and Nieuwland, 1995) or shown no decline in abundance from 1973 to 1992 (Bak et al. 2005), even though other non-agariciid corals on the same deep reefs have decreased. However, it is unknown whether this relative stability at depth holds across the full range of the species. As compared to the range-wide population estimates, the potential loss of 379 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for *Agaricia lamarcki* include the potential losses of this species to bleaching or disease (Brainard et al. 2011). When bleaching occurs for this species, effects can be severe; the species also likely has limited sediment tolerance. A factor that reduces extinction risk is that it occurs primarily at great depth, where disturbance events are less frequent. Despite low rates of sexual recruitment, the species is relatively persistent compared to other deep corals. The proposed project would not cause an increase in disease or bleaching. Therefore, NMFS believes that the proposed action is not likely to reduce the likelihood of Lamarck's sheet coral recovery in the wild.

Elliptical Star Coral

The proposed action will not affect the species' current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 16,465 colonies would result in a reduction in elliptical star coral distribution in the immediate action area. However, the species is found throughout the Caribbean, the Gulf of Mexico, Florida (including the Florida Middle Grounds), the Bahamas, and Bermuda (Brainard et al. 2011). In Florida, elliptical star coral has been recorded in the Florida Keys National Marine Sanctuary, Flower Garden Banks, National Marine Sanctuary, and Biscayne National Park. The action area for this project is located in the middle of this range. The proposed action will not result in a reduction of elliptical star coral distribution or fragmentation of the range since we expect that elliptical star coral will persist within the action area due to relocation of colonies (from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in elliptical star coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction. We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species. The anticipated loss of 16,465 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in elliptical star coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the elliptical star coral colonies occur in the smaller size classes and no corals were observed larger than 40 cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5-1 cm/yr) we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of elliptical star corals in the wild.

An estimated maximum of 16,465 colonies of elliptical star coral will be lethally taken during dredging activities. While we do not have exact population estimates for this species, a high number of colonies are believed to be still in existence through the species' range. The overall colony density of *Dichocoenia stokesi* averaged across all habitat types in the south Florida region was ~ 1.6 colonies per 10 m^2, making it the ninth most abundant coral species in this region (Wagner et al., 2010). A 2011 survey conducted by Nova Southeastern University just south of Port Everglades has identified 5,514 colonies of elliptical star coral over just 735 acres. As compared to the range-wide population estimates, the potential loss of 16,465 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for *Dichocoenia stokesi* include its documented population-level impacts from disease. The proposed project would not cause an increase in disease. Factors that reduce potential extinction risk are its relatively high abundance and persistence across many habitat types, including nearshore and mesophotic reefs. Residency in a wide range of habitat types suggests the species has a wide tolerance to environmental conditions and, therefore, better capacity to deal with changing environmental regimes. Therefore, NMFS believes that the proposed action is not likely to reduce the likelihood of elliptical star coral recovery in the wild.

Lobed Star Coral
The proposed action will not affect the species' current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 20,062 colonies of lobed star coral colonies would result in a reduction in lobed star coral distribution in the immediate action area. However, the species is common throughout U.S. waters of the western Atlantic and greater Caribbean, including Florida and the Gulf of Mexico. Within its range it is found within federally protected waters in the Flower Garden Bank Sanctuary, Dry Tortugas National Park, Virgin Islands National Park/Monument, Biscayne National Park, Florida Keys National Marine Sanctuary, Navassa National Wildlife Refuge, and the Buck Island Reef National Monument. The proposed action will not result in a reduction of lobed star coral distribution or fragmentation of the range since we expect that lobed star coral will persist within the action area due to relocation of colonies

(from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in lobed star coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction. We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species. The anticipated loss of 20,062 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in lobed star coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the lobed star coral colonies occur in the smaller size classes and no corals were observed larger than 40-cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5 -1 cm/yr) we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of lobed star corals in the wild.

While it is now widely accepted that *O. annularis* is only 1 of 3 valid species (the others being *O. franksi and O. faveolata*), long-term monitoring data sets and previous ecological studies did not distinguish among them, referring instead to the *Orbicella* complex. Although the biological review team that conducted the status review that resulted in the proposal to list these species estimated extinction risk separately for each species, much of the information available is for the complex as a whole (Brainard et al. 2011). An estimated maximum of 20,062 colonies of lobed star coral will be lethally taken during dredging activities. There is ample evidence that it has declined dramatically throughout its range (but perhaps at a slower pace than its fast-paced Caribbean colleagues, *Acropora palmata* and *Acropora cervicornis*). However, the *Orbicella* complex has historically been a dominant species on Caribbean and Florida coral reefs, characterizing the so-called "buttress zone" and "annularis zone" in the classical descriptions of Caribbean reefs (Goreau, 1959). Therefore, we believe that, even with the recent declines, there are still high numbers of lobed star coral throughout its range. As compared to the range-wide population estimates, the potential loss of 20,062 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for lobed star coral include very low productivity (growth and recruitment), documented dramatic declines in abundance, its restriction to the degraded reefs of the wider Caribbean region, and its preferential occurrence in shallow habitats. The proposed project would not increase any of these threats. Therefore, NMFS believes that the proposed action is not likely to reduce the likelihood of lobed star coral recovery in the wild.

Mountainous Star Coral
The proposed action will not affect the species' current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 627 colonies of mountainous star coral colonies would result in a reduction in mountainous star coral distribution in the immediate action area. However, the species is common throughout U.S. waters of the western Atlantic and greater Caribbean, including Florida and the Gulf of Mexico. Within its range it is found within federally protected waters in the Flower Garden Bank Sanctuary, Dry Tortugas National Park, Virgin Islands National Park/Monument, Biscayne National Park, Florida Keys National Marine Sanctuary, Navassa National Wildlife Refuge, and the Buck Island Reef National Monument. The proposed action will not result in a reduction of mountainous star coral distribution or fragmentation of the range since we expect that mountainous star coral will persist within the action area due to relocation of colonies (from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in mountainous star coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction. We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species. The anticipated loss of 627 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in mountainous star coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the mountainous star coral colonies occur in the smaller size classes and no corals were observed larger than 40-cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5 -1 cm/yr) we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of mountainous star corals in the wild.

While it is now widely accepted that *O. faveolata* is only 1 of 3 valid species (the others being *O. franksi and O. annularis*), long-term monitoring data sets and previous ecological studies did not distinguish among them, referring instead to the *Orbicella* complex. Although the biological review team has estimated extinction risk separately for each species, much of the information available is for the complex as a whole (Brainard et al. 2011). An estimated maximum of 627 colonies of mountainous star coral will be lethally taken during dredging activities. There is ample evidence that it has declined dramatically throughout its range (but perhaps at a slower pace than its fast-paced Caribbean colleagues, elkhorn and staghorn corals [*Acropora palmata* and *Acropora cervicornis*]). However, the *Orbicella* complex has historically been a dominant species on Caribbean and Florida coral reefs, characterizing the so-called “buttress zone” and “annularis zone” in the classical descriptions of Caribbean reefs (Goreau, 1959). Therefore, we believe that even with the recent declines that there are still high numbers of mountainous star coral throughout its range. A 2011 survey conducted by Nova Southeastern University just south of Port Everglades has identified 4,030 colonies of mountainous star coral over just 735 acres. As compared to the range-wide population estimates, the potential loss of 627 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for mountainous star coral include very low productivity (growth and recruitment), documented dramatic declines in abundance, its restriction to the degraded reefs of the wider Caribbean region, and its preferential occurrence in shallow habitats. The proposed project would not increase any of these threats. Therefore, NMFS believes that the proposed action is not likely to reduce the chances of mountainous star coral recovery in the wild.

Knobby Star Coral
The proposed action will not affect the species’ current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 627 colonies of knobby star coral colonies would result in a reduction in knobby star coral distribution in the immediate action area. However, the species is is common throughout U.S. waters of the western Atlantic and greater Caribbean, including Florida and the Gulf of Mexico. Within its range it is found within federally-protected waters in the Flower Garden Bank Sanctuary, Dry Tortugas National Park, Virgin Islands National Park/Monument, Biscayne National Park, Florida Keys National Marine Sanctuary, Navassa National Wildlife Refuge, and the Buck Island Reef National Monument. The proposed action will not result in a reduction of knobby star coral distribution or fragmentation of the range since we expect that knobby star coral will persist within the action area due to relocation of colonies (from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in knobby star coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction. We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species. The anticipated loss of 627 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in knobby star coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the knobby star coral colonies occur in the smaller size classes and no corals were observed larger than 40 cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5-1 cm/yr), we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of knobby star corals in the wild.

While it is now widely accepted that *O. franksi* is only 1 of 3 valid species (the others being *O. faveolata and O. annularis*), long-term monitoring data sets and previous ecological studies did not distinguish among them, referring instead to the *Orbicella* complex. Although the biological review team has estimated extinction risk separately for each species, much of the information available is for the complex as a whole (Brainard et al. 2011). An estimated maximum of 627 colonies of knobby star coral will be lethally taken during dredging activities. There is ample evidence that it has declined dramatically throughout its range (but perhaps at a slower pace than its fast-paced Caribbean colleagues, *Acropora palmata* and *Acropora cervicornis*). However, the *Orbicella* complex has historically been a dominant species on Caribbean and Florida coral reefs, characterizing the so-called "buttress zone" and "annularis zone" in the classical descriptions of Caribbean reefs (Goreau, 1959). Therefore, we believe that even with the recent declines that there are still high numbers of knobby star coral throughout its range. As compared to the range-wide population estimates, the potential loss of 627 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for knobby star coral include very low productivity (growth and recruitment), documented dramatic declines in abundance, its restriction to the degraded reefs of the wider Caribbean region, and its preferential occurrence in shallow habitats. The proposed project would not increase any of these threats. Therefore, NMFS believes that the proposed action is not likely to reduce the likelihood of knobby star coral recovery in the wild.

Rough Cactus Coral

The proposed action will not affect the species' current geographic range. Since relocated colonies will remain in the same area, no change in species distribution is anticipated. The anticipated mortalities of up to 1,207 colonies of rough cactus coral colonies would result in a reduction in rough cactus coral distribution in the immediate action area. However, *Mycetophyllia ferox* occurs throughout the U.S. waters of the western Atlantic but has not been reported from Flower Garden Banks (Hickerson et al., 2008). Within its range it is found within federally-protected waters in the Dry Tortugas National Park, Virgin Islands National Park/Monument, Biscayne National Park, Florida Keys National Marine Sanctuary, Navassa National Wildlife Refuge, and the Buck Island Reef National Monument. The proposed action will not result in a reduction of rough cactus coral distribution or fragmentation of the range since we expect that rough cactus coral will persist within the action area due to relocation of colonies (from the impact area to the artificial reef area) and will continue to be capable of reproducing. Therefore, the reproductive potential of the species in this portion of its range will persist.

Although no change in rough cactus coral distribution was anticipated, we concluded lethal takes would result in a reduction in absolute population numbers that may also reduce reproduction.

We believe these reductions are unlikely to appreciably reduce the likelihood of survival of the species in the wild, because the action will not negatively affect critical metrics of the status of the species. The anticipated loss of 1,207 colonies would reduce the population by that amount, compared to the number that would have been present in the absence of the proposed action, assuming all other variables remained the same. Therefore, the action will result in a reduction in rough cactus coral reproduction, but would not have a measurable effect on the distribution of the species within the Florida unit or throughout its range.

According to the resource surveys conducted by Dial Cordy, Inc., the majority of the rough cactus coral colonies occur in the smaller size classes and no corals were observed larger than 40-cm longest linear dimension. Reproductive potential is positively correlated with colony size. In the species for which we have estimates of size at first reproduction, all are larger than 40 cm (average ~100 cm). Thus, we assume that these corals are not currently reproductive. Further, given the relatively slow growth rates of the proposed corals (~0.5 -1 cm/yr), we do not anticipate that these colonies would reach reproductive maturity over the duration of the project (i.e., 5 years). Therefore, we believe that the proposed project will not result in a reduction in reproduction of rough cactus corals in the wild.

An estimated maximum of 1,207 colonies of rough cactus coral will be lethally taken during dredging activities. Mycetophyllia ferox is usually uncommon (Veron, 2000) or rare according to published and unpublished records, indicating that it constitutes < 0.1% species contribution (percent of all colonies censused) and occurs at densities < 0.8 colonies per 10 m^2 in Florida (Wagner et al., 2010) and at 0.8 colonies per 100 m transect in Puerto Rico sites sampled by the Atlantic and Gulf Rapid Reef Assessment (AGRRA database online at http://www.agrra.org). Recent monitoring data (e.g., since 2000) from Florida (National Park Service permanent monitoring stations), La Parguera (Puerto Rico), and St. Croix (USVI/NOAA Center for Coastal Monitoring and Assessment randomized monitoring stations) show *Mycetophyllia ferox* cover to be consistently less than 1%, with occasional observations up to 2% and no apparent temporal trend (available online at http://www8.nos.noaa.gov/biogeo_public/query_habitat.aspx). Given the amount of reef tract in Florida, even at <0.8 colonies per $10m^2$ there is still likely to be a high number of rough cactus coral colonies throughout Florida and even higher numbers throughout the range. As compared to the range-wide population estimates, the potential loss of 1,207 colonies would cause no noticeable change in the population of the species. Therefore, we believe the proposed action will not reduce appreciably the likelihood of survival in the wild.

Factors that increase the extinction risk for *Mycetophyllia ferox* include disease and rare abundance. Limited available information suggests that this species suffered substantial population declines in recent decades, primarily as a result of coral disease. The proposed project would not increase coral disease. Therefore, NMFS believes that the proposed action is not likely to reduce the likelihood of mountainous star coral recovery in the wild.

Staghorn Coral

The blended mitigation plan involves directed (intentional) take of coral fragments from up to 250 colonies of staghorn coral and collection of up to 2,500 “coral fragments of opportunity.”

Although a recovery plan has not been finalized at this time for staghorn coral, we consider the recovery vision statement from the *Acropora* Recovery Outline (available at http://sero.nmfs.noaa.gov/pr/protres.htm) relevant to analyze the effects on recovery:

> Staghorn (and elkhorn) coral populations should be large enough so that reproducing individuals comprise numerous populations across their historical geographic range (wider Caribbean) and additionally, should be large enough to protect the species' genetic diversity. Threats to the species and habitat loss and degradation will be sufficiently abated to ensure a high probability of survival into the future.

No reduction in numbers, reproductive potential, or distribution of staghorn coral will result from the proposed actions. The directed take through fragment collection from up to 250 wild staghorn colonies will result in temporary impacts to the donor colony and will not result in the removal of any whole colonies. Corals of opportunity collected for propagation and outplanting would have likely otherwise died since they are unattached to the seafloor (e.g., as a result of ship groundings or storm events). Staghorn fragments collected from either source will be propagated and outplanted to degraded reefs within the action area. Therefore, there will be no reduction in numbers of staghorn coral. Rather, through the proposed action, there will be an increase in numbers of staghorn coral.

The collection of small fragments from the branch tips of staghorn coral is not anticipated to have any effect on the sexual or asexual reproduction of the donor colonies. Coral fragments are not collected during the summer months when the corals are producing eggs and sperm. The growing tip heals quickly and regains its reproductive potential quickly also. As stated above, corals of opportunity would likely have died without collection; therefore, collection results in preservation of the reproductive potential of the fragment. Last, the collected fragments will be propagated and outplanted resulting in an increase of reproductive output as compared to the potential the fragment had prior to collection. NMFS does not believe the proposed action is likely to impede staghorn coral's ability to reproduce sexually due to the loss or impacts to the reproductive capacity.

The proposed action is not expected to result in any lethal take of staghorn coral. We have determined that the directed take through fragment collection from up to 250 wild staghorn colonies will result in temporary impacts to the donor colony and will not result in the removal of any whole colonies. Collection of corals of opportunity will not reduce the species range, as these colonies would have died without having been collected. Further, the collected fragments will be propagated within nurseries and outplanted within the same geographic area (Broward County). Therefore, we believe that there would be no measurable effect on the distribution of the species throughout its range. The proposed action will not result in a reduction of numbers, reproduction, or distribution of staghorn corals. Hence, we have determined that the proposed action is not expected to appreciably reduce the likelihood of survival and recovery of these coral species in the wild.

9 Analysis of Destruction or Adverse Modification of Designated Critical Habitat

Critical habitat was designated for elkhorn and staghorn corals, in part, because further declines in the low population sizes of the species could lead to threshold levels that make the chances for recovery low. More specifically, low population sizes for these species could lead to an Allee effect and lower effective density (of genetically distinct adults required for sexual reproduction), and a reduced source of fragments for asexual reproduction and recruitment. In other words, a staghorn coral mate may be too far away for successful sexual reproduction to occur. Therefore, the key conservation objective of designated critical habitat is to facilitate increased incidence of successful sexual and asexual reproduction (i.e., increase the potential for sexual and asexual reproduction to be successful), which in turn facilitates increases in the species' abundance, distribution, and genetic diversity. To this end, our analysis of whether the proposed action is likely to destroy or adversely modify designated critical habitat seeks to determine if the adverse effects of proposed action on the essential features of designated *Acropora* critical habitat will appreciably reduce the capability of the critical habitat to facilitate an increased incidence of successful sexual and asexual reproduction. This analysis takes into account the current status of each species; for example, the level of increased incidence of successful reproduction that needs to be facilitated may be different depending on the recovery status of elkhorn and staghorn corals in the action area. This analysis also takes into account the geographic and temporal scope of the proposed action, recognizing that functionality of critical habitat necessarily means that it must currently and in the future continue to support the conservation of the species and progress toward recovery.

The key objective for the conservation and recovery of listed coral species identified for the designated critical habitat is the facilitation of an increase in the incidence of sexual and asexual reproduction. Recovery cannot occur without protecting the essential feature of critical habitat from destruction or adverse modification because the quality and quantity of suitable substrate for listed corals affects their reproductive success. The proposed action will result in the permanent loss of up to 23.62 acres of critical habitat via direct removal, fracturing and rubble creation, and sedimentation. Therefore, this portion of critical habitat will be permanently unavailable and unsuitable for coral recruitment.

As described in Section 6.3 above, the permanently affected portion of the critical habitat has the conservation potential of supporting up to 31,878 colonies of staghorn coral. This conservation potential would be realized if the area was eventually occupied by that number of colonies, which is dependent on several biological and physical factors that affect future success of staghorn corals colonizing these particular reefs. At the time of the critical habitat designation, and given the severely decreased abundance and depressed sexual reproduction of staghorn and elkhorn corals, NMFS determined it was necessary to include all of the essential feature (i.e., settlement substrate) that occurs in each critical habitat area within the designation, to maximize the potential that successful recruitment could occur. However, on finer scales, not every portion of critical habitat has the exact same conservation potential at any given time. The critical habitat within the project area is subject to wave action and pollution from constant large vessel traffic (i.e., physical and chemical barriers to conservation potential of the critical habitat).

Furthermore, there are currently no visible staghorn colonies within or near the permanently affected critical habitat (Dial Cordy, Inc. 2006) which would be capable of providing gametes for sexual reproduction (i.e., there currently exists a biological barrier to the conservation potential of the critical habitat). Therefore, we believe that the critical habitat within the dredge footprint would not achieve the density of adult staghorn colonies necessary to facilitate sexual reproduction (i.e., the conservation potential) during the construction timeframe of the project (up to 10 years) and potentially not soon after construction, given the completed project may exacerbate the physical and chemical barriers to recruitment discussed above.

The blended mitigation plan will likely use existing permitted coral nurseries which could supply colonies of suitable size (20 cm or greater) immediately. Even if colonies were not immediately available, it would take less than 1 year to grow corals to the appropriate size and begin outplanting. In either case, outplanting is expected to be implemented several years before the project construction impacts would occur. The above calculations indicate the minimum number of staghorn colonies necessary to immediately attain and maintain the density and coverage area recommended to achieve the population-based goals identified by the recovery team, and therefore also, the number of colonies necessary to provide the maximum conservation potential of the permanently adversely affected critical habitat area. However, this assumes 100% survival of outplanted colonies and no partial mortality. The assumption that no mortality will occur is unrealistic based on observations of approximately 75%-90% survival of outplants. Therefore, we assume a 20% increase in the number of required outplanted colonies (20% of 31,878 = 6,376 additional colonies) to buffer against loss from sources of mortality such as disease, predation, and stochastic events. Thus, accounting for the number of colonies needed to meet the population goal and the number of colonies needed to buffer against mortality, 38,254 colonies (31,878 + 6,376 = 38,254) of the size and density described above, are necessary to fully realize the conservation potential of the permanently affected area.

As stated above, we do not believe that the critical habitat in the dredge footprint will meet the conservation goal within the lifetime of the project, if ever. The propagation and outplanting will achieve the maximum conservation benefit of appropriate numbers and densities of adult staghorn colonies within 7 years. Also, the number of staghorn colonies necessary to meet the conservation potential of the permanently affected area is at the low end of the range of staghorn coral colonies proposed in the mitigation plan (i.e., 35,000-50,000). We believe that despite the loss of 23.62 acres of habitat the proposed action will have beneficial effects on designated critical habitat, by accelerating the provision of its intended conservations functions for staghorn coral; the conservation potential of the critical habitat areas to be occupied by the outplanted colonies will be realized on a much shorter time scale than would be possible naturally due to the currently low population densities of staghorn within this portion of designated critical habitat.

Above we calculated the maximum reproductive potential of the adversely affected portion of critical habitat as supporting up to 31,878 of 20-cm-diameter colonies of staghorn coral at the densities and coverages identified by the recovery team. Thus, this portion of critical habitat would realize its full potential to support recovery if it were colonized by 31,878 colonies of staghorn coral. As noted above, we do not believe that this portion of critical habitat would reach that goal during the lifetime of this project, if ever, given the current depressed abundance of staghorn coral that could provide recruits to colonize the area, and the physical barriers to colonization resulting from the existence of the harbor. The proposed action also includes the propagation and outplanting of 35,000 to 50,000 20-cm staghorn corals as part of the USACE's mitigation plan. These colonies will be outplanted over a 7-year period, thus contributing to the recovery of the species within this portion of the species' range over a very short time frame. The propagation and outplanting of staghorn coral will exceed the reproductive potential of the portion of critical habitat adversely affected. Further, the final blended mitigation plan will require that the outplanting be conducted in densities and genotypic composition to maximize the sexual reproductive potential. Currently, the species' low population size and patchy distribution is impeding the chances of successful sexual reproduction. The outplanting will increase the sexual reproductive potential within this portion of critical habitat as compared to status quo. Therefore, the project as a whole will not impede the recovery of the listed corals in the action area or range-wide despite the loss of 23.62 acres of critical habitat within the action area. Rather, it may even increase the likelihood of recovery of the species. As such, the proposed project would not destroy or adversely modify the designated critical habitat for listed corals.

In the event that the USACE selects a contractor who will anchor outside of the channel we have determined that there will be an additional 19.31 acres of permanent adverse impacts to critical habitat from anchor and cable drag. These impacts are considered *potential* impacts at this time, because they may or may not occur depending on the contractor selected. Therefore, they were not included in the total calculations above for the blended mitigation plan requirements. Should an additional impact of up to 19.31 acres of critical habitat result from anchor placement and cable drag, additional reef enhancement will be required via coral propagation and outplanting. An appropriate amount of staghorn corals will be included to achieve the mitigation requirements and conservation of the species. The USACE will be required to ouplant staghorn corals of the size and density described above in order to realize the full conservation potential of the additional 19.31 acres of critical habitat that would be permanently lost.

10 Conclusion

Using the best available data, we analyzed the effects of the proposed action in the context of the status of the species, the environmental baseline, and cumulative effects, and determined that the proposed action is not likely to jeopardize the continued existence of staghorn coral, any of the 6 corals proposed for listing, or Johnson's seagrass. These analyses focused on the impacts to, and population responses of, these species. Because the proposed action will not reduce the likelihood of survival and recovery of corals proposed for listing or Johnson's seagrass, it is our opinion that the proposed action is also not likely to jeopardize the continued existence of these species.

After reviewing the current status of staghorn coral critical habitat, the environmental baseline, the effects of the proposed actions, and the cumulative effects, it is our opinion that the expansion of Port Everglades will not impede the critical habitat's ability to support the conservation of staghorn (or elkhorn) corals and therefore will not destroy or adversely modify the critical habitat.

11 Incidental Take Statement

Section 9 of the ESA and federal regulation pursuant to Section 4(d) of the ESA prohibit take of endangered and threatened species, respectively, without special exemption. *Take* is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. *Incidental take* is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of Section 7(b)(4) and Section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement (ITS).

NMFS must estimate the extent of take expected to occur from implementation of the proposed action to frame the limits of the take exemption provided in the Incidental Take Statement. These limits set thresholds that, if exceeded, would be the basis for reinitiating consultation. The following section describes the extent of take that NMFS anticipates will occur as a result of implementing the proposed action. If actual take exceeds an amount (or geographic or temporal extent) specified here, the exemption from the prohibition on take will be invalid for the excess amount, and re-initiation of consultation is required.

The prohibitions against taking the species found in Section 9 of the Act do not apply until the species is listed. However, NMFS advises the USACE to consider implementing the following reasonable and prudent measures. If the conference consultation on species proposed to be listed in opinion is adopted as a biological opinion following final listing, these measures, with their implementing terms and conditions, will be nondiscretionary, and must be undertaken by the USACE so that they become binding conditions of any grant, permit, or contract issued, as appropriate, for the exemption in Section 7(o)(2) to apply. The USACE has a continuing duty to regulate the activity covered by this incidental take statement. If the USACE (1) fails to assume and implement the terms and conditions or (2) fails to require the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. In order to monitor the impact of incidental take, the USACE must report the progress of the action and its impact on the species to NMFS as specified in the Incidental Take Statement (50 CFR §402.14(i)(3)).

11.1 Anticipated Amount of Take for Sea Turtles

Based on historical distribution data and hopper dredge observer take reports documenting previous hopper dredge takes of loggerhead and green sea turtles near the action area, we estimate that these 2 species may occur in the action area and may be taken by the hopper dredging operations of this project, by crushing and/or entrainment in suction dragheads. NMFS anticipates incidental take will consist of a total of 2 sea turtles (1 green and 1 loggerhead, or 2 greens, or 2 loggerheads) killed during hopper dredging at Port Everglades. Based on previous experience, we believe only 1 of these takes will be entrained, detected, and/or documented by onboard protected species observers. Therefore, we believe that there will be 1 *observed* take of either a green or a loggerhead.

Effect of the Take
NMFS has determined the anticipated level of incidental take specified in Section 11.1 is not likely to jeopardize the continued existence of loggerhead (NWA DPS), or green sea turtles.

11.2. Extent of Anticipated Take – Staghorn Coral

NMFS anticipates that the proposed action will result in take of this species in the form of collection. Coral nurseries will be established or augmented to support propagation of staghorn coral. Up to 250 colonies of staghorn corals are likely to be collected from the wild to supplement corals already established in nursery. Approximately 2,500 coral fragments of opportunity may also be collected. “Corals of opportunity” are the preferred source of colonies to populate coral nurseries, because they would have otherwise likely died without being collected and brought into the nursery. Sometimes, however, it is necessary to collect fragments from the wild to supplement the nursery population.

The protective regulations pursuant to ESA section 4(d) for staghorn provides for certain exceptions to the ESA section 9 prohibitions scientific research and species enhancement, and restoration carried out by authorized personnel (73 FR 64264; October 29, 2008). Thus, the take that may result from this project’s propagation and outplanting of staghorn corals is not prohibited, as long as the actions are carried out pursuant to: (1) the exceptions in the 4(d) rule and (2) the Biological Opinion on the issuance of the rule. Because all activities related to coral propagation (i.e., wild collection, nursery establishment and operation, and outplanting) in Broward County require a State of Florida Special Activity License (SAL), the USACE will be required to hold a valid permit (SAL) and they will ne incompliance with the 4(d) rule. Thus, the take will not be prohibited and no incidental take statement is required. However, see the discussion below regarding consequences of the potential reclassification of this species to endangered status.

11.3 Extent of Anticipated Take – Proposed Corals Including Staghorn Listed as Endangered

As previously stated, NMFS has proposed to reclassify staghorn coral from threatened to endangered and proposed to list 6 additional coral species that occur within the action area. Should that proposal become final (decision due June 2014), the aforementioned 4(d) rule for staghorn corals will be void because there are no exceptions to the take prohibitions allowed for endangered corals. Therefore, the take that will result from the propagation activities will need authorization. Further, 6 coral species proposed for listing will be either lethally taken or relocated from the action area. Therefore, the take that would result from these activities would also have to be authorized.

In additional to take being authorized through biological opinions, NMFS may authorize take through ESA Section 10. Take that results from scientific research or enhancement activities may be authorized by an ESA Section 10(a)(1)(A) permit. Because there are multiple staghorn coral nurseries currently operational, the NOAA Restoration Center has submitted an application for an ESA Section 10(a)(1)(A) permit for propagation of the proposed endangered corals, in anticipation and preparation for the potential that they may be listed as endangered. This permit application covers activities conducted by the Restoration Center and its partners. It is anticipated that one of these entities would carry out the coral propagation activities proposed in the Port Everglades expansion; thus, should they receive their ESA Section 10 permit, the take that would result from this project would be authorized. Should that permit not be issued by the time the propagation portion of this project commences, the USACE will need to reinitiate consultation to request authorization for the take.

Since the USACE has requested conference consultation on the proposed species, at the proper time they must request that this Conference Opinion be confirmed as NMFS's Biological Opinion should the species be listed/reclassified. At that time, the USACE will also request take authorization for the corals that are ultimately listed as endangered that are proposed to be lethally taken and/or relocated from the action area. Based on our analyses in Section 6.5, we anticipate the following take of the proposed corals:

Table 8. Estimated Maximum Amount of Take of Proposed Coral Species From the Port Everglades Expansion Project

Proposed Coral Species	Mortality (Middle and Outer Reef <57ft 15.55 ac)	Relocation Survival (Middle and Outer Reef 15.55 ac)	Relocation Mortality (Middle and Outer Reef 15.55 ac)	Mortality (Middle and Outer Reef >57ft 6.11 ac)	Mortality (Channel Bottom and Walls 133 ac)	Mortality (Indirect Impact Area 1.96 acres)	Mortality Total
Lamarck's sheet	0	35	6	16	352	5	379
Elliptical star	1522	105	19	646	14,071	207	16465
Lobed star	1121	773	657	792	17,238	254	20062
Mountainous star	36	25	21	29	24	517	627
Knobby star	36	25	21	29	24	517	627
Rough cactus	82	35	6	48	1,055	16	1207

Effect of the Take

NMFS has determined the anticipated take specified in Section 11.2 is not likely to jeopardize the continued existence of staghorn coral if the project is developed as proposed. NMFS also has determined the anticipated take specified in Section 11.3 is not likely to jeopardize the continued existence of any of the proposed corals if the project is developed as proposed.

12 Reasonable and Prudent Measures (RPMs)

Section 7(b)(4) of the ESA requires NMFS to issue a statement specifying the impact of any incidental take on listed species, which results from an agency action otherwise found to comply with Section 7(a)(2) of the ESA. It also states that the RPMs necessary to minimize the impacts of take and the terms and conditions to implement those measures must be provided and must be followed to minimize those impacts. Only incidental taking by the federal agency or applicant that complies with the specified terms and conditions is authorized.

The RPMs and terms and conditions are specified as required by 50 CFR 402.12 (i)(1)(ii) and (iv) to document the incidental take by the proposed action and to minimize the impact of that take on staghorn coral. These measures and terms and conditions are non-discretionary, and must be implemented by the USACE or the contractor in order for the protection of Section 7(o)(2) to apply. The USACE has a continuing duty to regulate the activity covered by this ITS. If the USACE or the contractor fails to adhere to the terms and conditions of the ITS through enforceable terms, and/or fails to retain oversight to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse. To monitor the impact of the incidental take, the USACE or the contractor must report the progress of the action and its impact on the species to NMFS as specified in the ITS [50 CFR 402.12(i)(3)].

NMFS has determined that the following RPMs are necessary and appropriate to minimize impacts of the incidental take of staghorn coral colonies and proposed coral species during the proposed action. The following RPMs and associated terms and conditions are established to implement these measures, and to document incidental takes. Only incidental takes that occur while these measures are in full implementation are authorized. These restrictions remain valid until reinitiation and conclusion of any subsequent Section 7 consultation.

1. Pre-construction survey. The USACE will conduct a pre-construction survey to document all listed and proposed species prior to construction.

2. The USACE must ensure that all colonies of coral species proposed to be listed that are over 10 centimeters are relocated from the middle and outer reefs prior to beginning construction. The USACE is also authorized to relocate smaller colonies (4 cm and greater). (Please note: the requirement to relocate corals of 10 cm or greater is based on specific details associated with this project and may not to be used as the standard for future biological opinions. NMFS recommends that the USACE relocate all proposed species greater than 4 cm.)

3. Blended Coral Mitigation plan. The USACE must refine and implement the blended mitigation plan discussed throughout this opinion.

4. Environmental monitoring plan. USACE must conduct environmental monitoring to assess whether environmental impacts of the project exceed thresholds identified in the DEIS.

The USACE must provide NMFS with all data collected during monitoring events conducted, as well as any monitoring reports generated following the completion of the proposed project. The monitoring programs shall include reporting requirements to ensure NMFS, USACE, and other relevant agencies are aware of corrective actions being taken when thresholds are exceeded, as well as ensure NMFS receives data related to the condition of listed corals in the area due to the importance of these listed species.

13 Terms and Conditions

In order to be exempt from liability for take prohibited by Section 9 of the ESA, USACE must comply with the following terms and conditions, which implement the RPMs described above. These terms and conditions are nondiscretionary.

1. USACE must record the location and size of all listed and proposed corals during the pre-construction surveys and provide this info to NMFS. (RPM 1)

2. Relocation of proposed coral species: Since transplantation can be stressful on corals and the natural environment is variable, we believe the best way to minimize stress and ensure the survival of all transplanted colonies is to follow the established protocols (see Appendix B). Qualified individuals following the protocols in Appendix B must conduct transplantation. The USACE must ensure that all transplanted colonies are relocated to suitable habitat near their original location, but no closer than 400 ft from the edge of the channel. For the purposes of this opinion, suitable habitat is considered: similar depth as origin (+/- 5ft), uncolonized hard substrate, appropriate water quality (based on water quality data and local knowledge), and minimal chances of other disturbances (boat groundings, damage caused by curious divers/fisherman). (RPM 2)

3. USACE must record the original location of each transplanted colony, as well as the location of each colony after transplantation. (RPM 2).

4. The detailed blended coral mitigation plan will continue to be refined and implemented in coordination with NMFS. The plan includes a comprehensive monitoring plan for all relocated and outplanted corals. USACE will submit a final, detailed mitigation plan to NMFS prior to construction. USACE will report progress in implementing and monitoring the mitigation plan, as specified in the final mitigation plan (RPM 3).

5. USACE shall continue to work with the established interagency team including USACE, NMFS, EPA, FDEP, and FWC to refine the environmental monitoring plan and to evaluate its effectiveness during implementation. USACE shall submit the final refined environmental monitoring plan to NMFS prior to construction. (RPM 4).

6. The monitoring methods employed shall be capable of detecting sedimentation and turbidity and physical impacts in coral reef and hardbottom habitat within 150 meters of the dredging areas, and detecting whether impacts are likely to exceed adverse impacts considered in this Opinion in a timely manner allowing for adaptive management, during all phases of construction. USACE shall share monitoring results with the interagency team (RPM 4).

7. In the event that monitoring of coral reef and hardbottom habitat within the 150 meter zone indicates that listed coral species are likely to be adversely impacted by dredging-related turbidity, sedimentation, or physical impacts in a manner or to a degree that would exceed the adverse impacts considered in this Opinion, USACE shall implement an adaptive management plan to avoid or minimize the impacts, which may include additional transplanting and monitoring of corals and hardbottom organisms. In developing the adaptive management plan, USACE shall consult with the interagency team and consider recommendations from the team. The USACE's selected adaptive management plan shall be provided to the interagency team before the time the adverse impacts considered in this opinion are expected to be exceeded. A goal of the adaptive management plan will be to avoid the need for reinitiation of consultation on this Opinion, but additional coordination may be required with NMFS to, for example, provide authorization for additional transplanting and relocation. (RPM 4).

8. USACE must ensure that all appropriate natural resource permits are obtained prior to relocation, propagation, and outplanting of corals. (RPMs 2, 3, and 4)

14 Conservation Recommendations

Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

NMFS believes the following conservation recommendations further the conservation of listed and proposed coral species. NMFS strongly recommends that these measures be considered and implemented, and requests to be notified of their implementation.

1. NMFS recommends that in addition to the proposed sharing of monitoring and reporting data, the USACE provide NMFS's Southeast Region Protected Resource Division (PRD), with the collected data submitted for all projects permitted concerning listed and proposed coral species.

2. NMFS continues to recommend that the USACE relocated all proposed and listed coral species 4 cm and greater. This is consistent with the best available science and the NMFS Habitat Office's conservation recommendations.

3. NMFS recommends that the USACE provide the location and size of all proposed and listed corals to all persons who hold the proper permits and who may be interested in rescuing those corals for use in research or educational activities.

4. NMFS strongly recommends that the USACE, in consultation with PRD, utilize its authority to carry out programs for the conservation of listed and proposed corals. Pursuant to ESA Section 7(a)(1), the USACE should develop a program to donate a

fragment of each acroporid colony directly impacted by all authorized or permitted activities to an appropriate coral nursery.

5. NMFS recommends that USACE prepare and use a report of all current and proposed USACE projects in the range of Johnson's seagrass to assess impacts on the species from these projects, to assess cumulative impacts, and to assist in early consultation that will avoid and/or minimize impacts to Johnson's seagrass and its critical habitat. Information in this report should include location and scope of each project and identify the federal lead agency for each project.

6. NMFS recommends that the USACE conduct and support research to assess trends in the distribution and abundance of Johnson's seagrass. USACE should contribute data collected to the Florida Fish and Wildlife Conservation Commission's Florida Wildlife Research Institute to support ongoing GIS mapping of Johnson's and other seagrass distribution.

7. NMFS recommends that the USACE, in coordination with seagrass researchers and industry, support ongoing research on light requirements and transplanting techniques to preserve and restore Johnson's seagrass, and on collection of plants for genetics research, tissue culture, and tissue banking.

8. NMFS recommends that the USACE prepare an assessment of the effects of other actions under its purview on Johnson's seagrass for consideration in future consultations.

9. NMFS recommends that the USACE promote the use of the October 2002, *Key for Construction Conditions for Docks or other Minor Structures Constructed in or over Johnson's Seagrass* as the standard construction methodology for proposed docks located in the range of Johnson's seagrass.

10. NMFS recommends that the USACE review and implement the recommendations in the July 2008 report, *The Effects of Docks on Seagrasses, With Particular Emphasis on the Threatened Seagrass, Halophila johnsonii* (Landry et al. 2008).

11. NMFS recommends that the USACE review and implement the Conclusions and Recommendations in the October 2008 report, *Evaluation of Regulatory Guidelines to Minimize Impacts to Seagrasses from Single-Family Residential Dock Structures in Florida and Puerto Rico* (Shafer et al. 2008).

In order to keep NMFS informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, NMFS requests notification of the implementation of any conservation recommendations.

15 Reinitiation of Consultation

As provided in 50 CFR Section 402.16, reinitiation of formal consultation is required where discretionary federal agency involvement or control over the action has been retained (or is authorized by law) and if (1) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not previously considered, (2) the identified action is subsequently modified in a manner that causes an effect to listed species or critical habitat that was not considered in the biological opinion, or (3) a new species is listed or critical habitat designated that may be affected by the identified action. In addition, if the USACE chooses a contractor or dredging methodology which may result in impacts to listed species or critical habitat above that which is considered in this Opinion, in particular, the impacts to critical habitat from anchoring discussed in section 6.3 above, reinitiation will be required.

16 Literature Cited

Abal, E. G., N. Loneragan, P. Bowen, C. Perry, J. Udy, and W. C. Dennison. 1994. Physiological and morphological responses of the seagrass Zostera capricorni Aschers to light intensity. Journal of Experimental Marine Biology and Ecology 178:113–129.

Acevedo, R., J. Morelock, and R. A. Olivieri. 1989. Modification of coral reef zonation by terrigenous sediment stress. Palaios 4(1):92-100.

Acosta, A., and A. Acevedo. 2006. Population structure and colony condition of *Dendrogyra cylindrus* (Anthozoa: Scleractinia) in Providencia Island, Columbian Caribbean. Pages 1605-1610 *in* Proceedings of the 10th International Coral Reef Symposium, Okinawa, Japan.

Acropora Biological Review Team. 2005. Atlantic *Acropora* Status Review Document. Report to National Marine Fisheries Service, Southeast Regional Office. March 3, 2005. 152 p. + App.

Adey, W.H. 1977. Shallow water Holocene bioherms of the Caribbean Sea and West Indies. Proceedings of the 3rd International Coral Reef Symposium 2: xxi-xxiii.

Adey, W.H. 1978. Coral reef morphogenesis: A multidimensional model. Science 202: 831-837.

Aeby, G. S., and D. L. Santavy. 2006. Factors affecting susceptibility of the coral *Montastraea faveolata* to black-band disease. Marine Ecology Progress Series 318:103-110.

Albright, R., B. Mason, and C. Langdon. 2008. Effect of aragonite saturation state on settlement and post-settlement growth of Porites astreoides larvae. Coral Reefs 27(3):485-490.

Albright, R., B. Mason, M. Miller, and C. Langdon. 2010. Ocean acidification compromises recruitment success of the threatened Caribbean coral *Acropora palmata*. Proceedings of the National Academy of Sciences 107(47):20400-20404.

Alvarado-Chacon, E. M., and A. Acosta. 2009. Population size-structure of the reef-coral *Montastraea annularis* in two contrasting reefs of a marine protected area in the southern Caribbean Sea. Bulletin of Marine Science 85(1):61-76.

Alvarez-Filip, L., N. K. Dulvy, J. A. Gill, I. M. Côté, and A. R. Watkinson. 2009. Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. Proceedings of the Royal Society B: Biological Sciences 276(1669):3019-3025.

Amos, A. F. 1989. The occurrence of hawksbills Eretmochelys imbricata along the Texas coast. Pages 9-11 in S.A. Eckert, K.L. Eckert, and T.H. Richardson, compilers. Proceedings of the ninth annual workshop on sea turtle conservation and biology, NOAA technical memorandum NMFS/SEFC-232.

Anthony, K. R. N., and coauthors. 2011. Ocean acidification and warming will lower coral reef resilience. Global Change Biology 17:1798–1808.

Anthony, K. R. N., and O. Hoegh Guldberg. 2003. Variation in coral photosynthesis, respiration and growth characteristics in contrasting light microhabitats: an analogue to plants in forest gaps and understoreys? Functional Ecology 17(2):246-259.

Aronson, R. B., and W. F. Precht. 2000. Herbivory and algal dynamics on the coral reef at Discovery Bay, Jamaica. Limnology and Oceanography 45(1):251-255.

Aronson, R. B., and W. F. Precht. 2001. White-band disease and the changing face of Caribbean coral reefs. Hydrobiologia 460(1):25-38.

Aronson, R. B., W. F. Precht, and I. G. Macintyre. 1998. Extrinsic control of species replacement on a Holocene reef in Belize: the role of coral disease. Coral Reefs 17(3):223-230.

Aronson, R., A. Bruckner, J. Moore, W. Precht, and E. Weil. 2008. IUCN Red List of Threatened Species. Pages IUCN 2012. IUCN Red List of Threatened Species. Version 2012.2 *in*. IUCN.

Babcock, R.C. 1991. Comparative demography of three species of Scleractinian corals using age- and size-dependent classifications. Ecological Monographs, 61:225–244.

Baggett, L. S., and T. J. Bright. 1985. Coral recruitment at the East Flower Garden Reef (Northwestern Gulf of Mexico). Pages 379-384 *in* Proceedings 5th International Coral Reef Congress, volume 4, Tahiti, Polynesia.

Baird, A. H., J. R. Guest, and B. L. Willis. 2009. Systematic and biogeographical patterns in the reproductive biology of scleractinian corals. Annual Review of Ecology, Evolution, and Systematics 40:531-571.

Bak, R. P. M. 1978. Lethal and sublethal effects of dredging on reef corals. Marine Pollution Bulletin 9(1):14-16.

Bak, R. P. M., and B. E. Luckhurst. 1980. Constancy and change in coral reef habitats along depth gradients at Curaçao. Oecologia 47(2):145-155.

Bak, R. P. M., and G. Nieuwland. 1995. Long-term change in coral communities along depth gradients over leeward reefs in the Netherlands Antilles. Bulletin of Marine Science 56(2):609-619.

Bak, R. P. M., and J. H. B. W. Elgershuizen. 1976. Patterns of oil-sediment rejection in corals. Marine Biology 37(2):105-113.

Bak, R. P. M., and M. S. Engel. 1979. Distribution, abundance and survival of juvenile hermatypic corals (Scleractinia) and the importance of life history strategies in the parent coral community. Marine Biology 54(4):341-352.

Bak, R. P. M., and S. R. Criens. 1982. Survival after fragmentation of colonies of *Madracis mirabilis*, *Acropora palmata* and *A. cervicornis* (Scleractinia) and the subsequent impact of a coral disease. Pages 221-227 *in* E. D. Gomez, and coeditors, editors. Proceedings of the Fourth International Coral Reef Symposium, Manila, Philippines.

Bak, R. P. M., G. Nieuwland, and E. H. Meesters. 2005. Coral reef crisis in deep and shallow reefs: 30 years of constancy and change in reefs of Curaçao and Bonaire. Coral Reefs 24(3):475-479.

Bak, R.P.M. 1977. Coral reefs and their zonation in the Netherland Antilles. AAPG Stud Geol. 4: 3-16.

Bak, R.P.M., Criens, S.R. 1982. Survival after fragmentation of colonies of *Madracis mirabilis*, *Acropora palmata* and *A. cervicornis* (Scleractinia) and the subsequent impact of a coral disease. Proceedings of the 4th International Coral Reef Symposium 1: 221-227.

Bak, R.P.M., Engel, M. 1979. Distribution, abundance and survival of juvenile hermatypic corals (Scleractinia) and the importance of life history strategies in the parent coral community. Marine Biology 54: 341-352.

Baker, J. D., C. L. Littnan, and D. W. Johnston. 2006. "Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna on the Northwestern Hawaiian Islands." Endangered Species Research 2:21-30.

Balazs, G. 1982. Growth rates of immature green turtles in the Hawaiian Archipelago. Biology and Conservation of Sea Turtles. K. A. Bjorndal. Washington D.C., Smithsonian Institution Press**:** 117-125.

Balazs, G. H. 1983. Recovery records of adult green turtles observed or originally tagged at French Frigate Shoals, northwestern Hawaiian Islands. Washington, D.C.; Springfield, VA, NMFS.

Balazs, G. H. 1985. Impact of ocean debris on marine turtles: entanglement and ingestion. Proceedings of the workshop on the fate and impact of marine debris, Honolulu, HI, NOAA-NMFS.

Bates, N. R., A. Amat, and A. J. Andersson. 2009. The interaction of ocean acidification and carbonate chemistry on coral reef calcification: evaluating the carbonate chemistry Coral Reef Ecosystem Feedback (CREF) hypothesis on the Bermuda coral reef. Biogeosciences Discussions 6:7627-7672.

Baums, I. B., C. R. Hughes, and M. E. Hellberg. 2005a. Mendelian microsatellite loci for the Caribbean coral Acropora palmata. Marine Ecology Progress Series 288:115-127.

Baums, I. B., M. E. Johnson, M. K. Devlin-Durante, and M. W. Miller. 2010. Host population genetic structure and zooxanthellae diversity of two reef-building coral species along the Florida Reef Tract and wider Caribbean. Coral Reefs 29:835–842.

Baums, I. B., M. W. Miller, and A. M. Szmant. 2003. Ecology of a corallivorous gastropod, Coralliophila abbreviata, on two scleractinian hosts. II. Feeding, respiration and growth. Marine Biology 142(6):1093-1101.

Baums, I. B., M. W. Miller, and M. E. Hellberg. 2005b. Regionally isolated populations of an imperiled Caribbean coral, Acropora palmata. Molecular Ecology 14(5):1377-1390.

Baums, I. B., M. W. Miller, and M. E. Hellberg. 2006. Geographic variation in clonal structure in a reef-building Caribbean coral, *Acropora palmata*. Ecological Monographs 76(4):503-519.

Beal, J.L., and Schmit. 2000. The effects of dock height on light irradiance (PAR) and seagrass (Halodule wrightii and Syringodium filiforme) cover, pp. 49-63. In: S.A. Bortone (ed) Seagrasses: Monitoring, Ecology, Physiology, and Management. Boca Raton, Florida. CRC Press.

Becker, L. C., and E. Mueller. 2001. The culture, transplantation and storage of Montastraea faveolata, Acropora cervicornis and Acropora palmata: What we have learned so far. Bulletin of Marine Science 69(2):881-896.

Bielmyer, G. K., and coauthors. 2010. Differential effects of copper on three species of scleractinian corals and their algal symbionts (Symbiodinium spp.). Aquatic Toxicology 97(2):125-133.

Birkeland, C. 1977. The importance of rate of biomass accumulation in early successional stages of benthic communities to the survival of coral recruits. Proceedings of the 3rd International Coral Reef Symposium 1: 15-21.

Birkeland, C. 1982. Terrestrial runoff as a cause of outbreaks of *Acanthaster planci* (Echinodermata: Asteroidea). Marine Biology 69(2):175-185.

Birrell, C. L., L. J. McCook, and B. L. Willis. 2005. Effects of algal turfs and sediment on coral settlement. Marine Pollution Bulletin 51(1-4):408-414.

Bjorndal, K. A. 1982. "The consequences of herbivory for the life history pattern of the Caribbean green turtle, Chelonia mydas. Pages 111-116 In: Bjorndal, K.A. (editor). Biology and Conservation of Sea Turtles." Smithsonian Institution Press. Washington, D.C.

Bjorndal, K. A. 1997. Foraging ecology and nutrition of sea turtles. The Biology of Sea Turtles. P. L. Lutz and J. A. Musick. Boca Raton, CRC Press.

Bjorndal, K. A., A. B. Bolten and M. Y. Chaloupka 2005. "Evaluating trends in abundance of immature green turtles, Chelonia mydas, in the Greater Caribbean." Ecological Applications **15**(1): 304-314.

Bjorndal, K. A., A. B. Bolten and Southeast Fisheries Science Center (U.S.) 2000. Proceedings of a workshop on Assessing Abundance and Trends for In-Water Sea Turtle Populations : held at the Archie Carr Center for Sea Turtle Research University of Florida, Gainesville, Florida, 24-26 March 2000. Miami, FL, U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Bjorndal, K. A., J. A. Wetherall, A. B. Bolten and J. A. Mortimer 1999. "Twenty-Six Years of Green Turtle Nesting at Tortuguero, Costa Rica: An Encouraging Trend." Conservation Biology 13(1): 126-134.

Bolten, A. B. and B. E. Witherington 2003. Loggerhead sea turtles. Washington, D.C., Smithsonian Books.

Bolten, A. B., K. A. Bjorndal and H. R. Martins. 1994. Life history model for the loggerhead sea turtle (Caretta caretta) populations in the Atlantic: Potential impacts of a longline fishery. NOAA Technical Memo, U.S. Department of Commerce.

Bolten, A. B., K. A. Bjorndal, H. R. Martins, T. Dellinger, M. J. Biscoito, S. E. Encalada and B. W. Bowen. 1998. "Transatlantic developmental migrations of loggerhead sea turtles demonstrated by mtDNA sequence analysis." Ecological Applications 8: 1-7.

Borger, J. L., and S. C. C. Steiner. 2005. The spatial and temporal dynamics of coral diseases in Dominica, West Indies. Bulletin of Marine Science 77(1):137-154.

Brainard, R. E., and coauthors. 2011a. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. U.S. Dep. Commer.

Brainard, R. E., and coauthors. 2011b. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. U.S. Dep. Commer.

Brandt, M. E. 2009. The effect of species and colony size on the bleaching response of reef-building corals in the Florida Keys during the 2005 mass bleaching event. Coral Reefs 28(4):911-924.

Brandt, M. E., and J. W. McManus. 2009. Disease incidence is related to bleaching extent in reef-building corals. Ecology 90(10):2859-2867.

Bright, A. J., D. E. Williams, K. L. Kramer, and M. W. Miller. 2013. Recovery of Acropora Palmata in Curacao: a Comparison with the Florida Keys. Bulletin of Marine Science 89(3):747-757.

Bruckner, A. W. 2002b. Proceedings of the Caribbean Acropora workshop: Potential application of the U.S. Endangered Species Act as a conservation strategy, volume 24. NOAA Office of Protected Resources, Silver Spring, MD.

Bruckner, A. W., and R. J. Bruckner. 1997. Outbreak of coral disease in Puerto Rico. Coral Reefs 16(4):260.

Bruckner, A. W., and R. J. Bruckner. 2006. Consequences of yellow band disease (YBD) on Montastraea annularis (species complex) populations on remote reefs off Mona Island, Puerto Rico. Diseases of Aquatic Organisms 69(1):67-73.

Bruckner, A. W., and R. L. Hill. 2009. Ten years of change to coral communities off Mona and Desecheo Islands, Puerto Rico, from disease and bleaching. Diseases of Aquatic Organisms 87(1-2):19-31.

Bruckner, A. W., editor. 2002. Proceedings of the Caribbean Acropora Workshop: Potential Application of the U.S. Endangered Species Act as a Conservation Strategy. NOAA Technical Memorandum NMFS-OPR-24, Silver Spring, MD.

Bruckner, A., R. Bruckner, and P. Sollins. 2000. Parrotfish predation on live coral:"spot biting" and "focused biting." Coral Reefs 19(1):50-50.

Bruno, J. F., and coauthors. 2007. Thermal stress and coral cover as drivers of coral disease outbreaks. PLoS Biology 5(6):e124.

Bruno, J. F., L. E. Petes, C. Drew Harvell, and A. Hettinger. 2003. Nutrient enrichment can increase the severity of coral diseases. Ecology Letters 6(12):1056-1061.

Bruno, J.F. 1998. Fragmentation in *Madracis mirabilis* (Duchassaing and Michelotti): how common is size-specific fragment survivorship in corals? J. Exp. Mar. Biol. Ecol., 230:169–181.

Budd, A. F., H. Fukami, N. D. Smith, and N. Knowlton. 2012. Taxonomic classification of the reef coral family Mussidae (Cnidaria: Anthozoa: Scleractinia). Zoological Journal of the Linnean Society 166(3):465-529.

Bulthuis, D. A. 1983. Effects of in situ light reduction on density and growth of the seagrass Heterozostera tasmanica (Martens ex Aschers.) den Hartog in Western Port, Victoria, Australia. Journal of Experimental Marine Biology and Ecology 67:91–103.

Burdick, D.M. and F.T. Short. 1999. The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. Environmental Management 23:231-240.

Burkepile, D. E., and M. E. Hay. 2007. Predator release of the gastropod *Cyphoma gibbosum* increases predation on gorgonian corals. Oecologia 154(1):167-173.

Cairns, S.D. 1982. Stony corals (Cnidaria: Hydrozoa, Scleractinia) of Carrie Bow Cay, Belize. *In*: Rutzler K, I.G. Macintyre (eds). The Atlantic barrier reef ecosystem at Carrie Bow Cay, Belize. Structure and communities. Smithson Contributions in Marine Science 12: 271-302.

Caldeira, K., and M. E. Wickett. 2003. Anthropogenic carbon and ocean pH. Nature 425(6956):365-365.

Caldow, C., and coauthors. 2009. Biogeographic characterization of fish communities and associated benthic habitats within the Flower Garden Banks National Marine Sanctuary, Silver Spring, MD.

CALTRANS. 2009. Technical Guidance for Assessment and Mitigation of the Hydroacoustic Effects of Pile Driving on Fish, Sacremento.

Campbell, C. L. and C. J. Lagueux 2005. "Survival probability estimates for large juvenile and adult green turtles (Chelonia mydas) exposed to an artisanal marine turtle fishery in the western Caribbean." Herpetologica 61(2).

Carilli, J. E., R. D. Norris, B. A. Black, S. M. Walsh, and M. McField. 2009. Local stressors reduce coral resilience to bleaching. PLoS ONE 4(7):e6324.

Carilli, J. E., R. D. Norris, B. Black, S. M. Walsh, and M. McField. 2010. Century-scale records of coral growth rates indicate that local stressors reduce coral thermal tolerance threshold. Global Change Biology 16(4):1247-1257.

Carpenter, K. E., and coauthors. 2008. One-Third of Reef-Building Corals Face Elevated Extinction Risk from Climate Change and Local Impacts. Science 321(5888):560-563.

Carricart-Ganivet, J. P., and M. Merino. 2001. Growth responses of the reef-building coral Montastraea annularis along a gradient of continental influence in the southern Gulf of Mexico. Bulletin of Marine Science 68(1):133-146.

Cervino, J. M., and coauthors. 2004. Relationship of Vibrio species infection and elevated temperatures to yellow blotch/band disease in Caribbean corals. Applied and Environmental Microbiology 70(11):6855-6864.

Chiappone, M. 2010. Public comment submitted to NMFS Southeast Regional Office, April 2010.

Chiappone, M., Sullivan, K.M. 1996. Distribution, abundance and species composition of juvenile scleractinian corals in the Florida reef tract. Bulletin of Marine Science 58: 555-569.

Clark, R., C. Jeffrey, K. Woody, Z. Hillis-Starr, and M. Monaco. 2009. Spatial and temporal patterns of coral bleaching around Buck Island Reef National Monument, St. Croix, US Virgin Islands. Bulletin of Marine Science 84(2):167-182.

Cohen, A. L., D. C. McCorkle, and S. de Putron. 2007. The impact of seawater saturation state on early skeletal development in larval corals: insights into scleractinian biomineralization. Proceedings of the American Geophysical Union 2007 Fall Meeting.

Cohen, A. L., D. C. McCorkle, S. de Putron, G. A. Gaetani, and K. A. Rose. 2009. Morphological and compositional changes in the skeletons of juvenile corals reared in acidified seawater: Insights into the biomineralization response to ocean acidification. Geochemistry Geophysics Geosystems 10:Q07005.

Cole, A. J., M. S. Pratchett, and G. P. Jones. 2008. Diversity and functional importance of coral-feeding fishes on tropical coral reefs. Fish and Fisheries 9(3):286-307.

Colella, M. A., R. R. Ruzicka, J. A. Kidney, J. M. Morrison, and V. B. Brinkhuis. 2012. Cold-water event of January 2010 results in catastrophic benthic mortality on patch reefs in the Florida Keys. Coral Reefs.

Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs. Science 199(4335):1302-1310.

Connell, J. H., T. P. Hughes, and C. C. Wallace. 1997. A 30-year study of coral abundance, recruitment, and disturbance at several scales in space and time. Ecological Monographs 67(4):461-488.

Connell, J.H. 1973. Population ecology of reef-building corals. *In*: Jones, O.A. and R. Endean (eds.). Biology and Geology of Coral Reefs, v. 2 pp 125-151. Corsolini, S., S. Aurigi and S. Focardi (2000). "Presence of polychlobiphenyls (PCBs) and coplanar congeners in the tissues of the Mediterranean loggerhead turtle Caretta caretta." Marine Pollution Bulletin 40: 952–960.

Correa, A. M. S., M. E. Brandt, T. B. Smith, D. J. Thornhill, and A. C. Baker. 2009. Symbiodinium associations with diseased and healthy scleractinian corals. Coral Reefs 28(2):437-448.

Craik, W., R. Kechington, and G. Kelleher. 1990. Coral-Reef Management. *In*: Z. Dubinsky (ed.). Ecosystems of the world coral reefs. Elsevier Science Publishers, NY pp.453-467.

Croquer, A., and coauthors. 2006. First report of folliculinid ciliates affecting Caribbean scleractinian corals. Coral Reefs 25(2):187-191.

Dallmeyer, D. G., J. W. Porter, and G. J. Smith. 1982. Effects of particulate peat on the behavior and physiology of the Jamaican reef-building coral Montastrea annularis. Marine Biology 68(3):229-233.

Dammann, A.E., and Nellis, D.W. 1992. A Natural History Atlas to the Cays of the U.S. Virgin Islands. Pineapple Press, Inc., Sarasota, FL. 160 pp.

Danovaro, R., and coauthors. 2008. Sunscreens cause coral bleaching by promoting viral infections. Environmental Health Perspectives 116(4):441-447.

Davis, G. E. 1977. Anchor damage to a coral reef on the coast of Florida. Biological Conservation 11(1):29-34.

Davis, G. E. 1982. A century of natural change in coral distribution at the Dry Tortugas: A comparison of reef maps from 1881 and 1976. Bulletin of Marine Science 32(2):608-623.

Dawes, C.J., C.S. Lobban, and D.A. Tomasko. 1989. A comparison of the physiological ecology of the seagrasses Halophila decipiens Ostenfeld and Halophila johnsonii Eiseman from Florida. Aquatic Botany 33:149-154.

De Putron, S., D. McCorkle, A. Cohen, and A. Dillon. 2010. The impact of seawater saturation state and bicarbonate ion concentration on calcification by new recruits of two Atlantic corals. Coral Reefs:1-8.

Debrot, A. O., M. M. C. E. Kuenen, and K. Dekker. 1998. Recent declines in the coral fauna of the Spaanse Water, Curaçao, Netherlands Antilles. Bulletin of Marine Science 63(3):571-580.

Delvoye, L. 1988. Gametogenesis and gametogenic cycles in Agaricia agaricites (L) and Agaricia humilis Verrill and notes on gametogenesis in Madracis mirabilis (Duchassaing & Michelotti) (Scleractinia). Foundation for Scientific Research in Surinam and the Netherlands Antilles 123:101-134.

Dennison, W. C. 1987. Effects of light on seagrass photosynthesis, growth, and depth distribution. Aquatic Botany 27:15–26.

DHSMV. 2010. Florida Department of Highway Safety and Motor Vehicles, Florida Vessel Owners: Statistics. Available at: http://www.flhsmv.gov/dmv/vslfacts.html

DiCarlo, G., F. Badalamenti, A.C. Jensen, E.W. Koch, and S. Riggio. 2005. Colonization process of vegetative fragments of Posidonia oceanica (L.) Delile on rubble mounds. Marine Biology 147:1261-1270.

Diez, C. E. and R. P. v. Dam 2002. "Habitat effect on hawksbill turtle growth rates on feeding grounds at Mona and Monito Islands, Puerto Rico." Marine Ecology Progress Series 234: 301-309.

Diez, C. E. and R. P. van Dam 2007. "In-water surveys for marine turtles at foraging grounds of Culebra Archipelago, Puerto Rico Progress Report: FY 2006-2007."

Doney, S., V. Fabry, R. Feely, and J. Kleypas. 2009. Ocean acidification: the other CO_2 problem. Annual Review of Marine Science 1.

Downs, C. A., and coauthors. 2005. Cellular diagnostics and coral health: Declining coral health in the Florida Keys. Marine Pollution Bulletin 51(5-7):558-569.

Dunne, R.P., Brown, B.E. 1979. Some aspects of the ecology of reefs surrounding Anegada, British Virgin Islands. Atoll Research Bulletin 236: 1-83.

Dupont, J., W. Jaap, and P. Hallock. 2008. A retrospective analysis and comparative study of stony coral assemblages in Biscayne National Park, FL (1977–2000). Caribbean Journal of Science 44(3):334–344.

Durako, M.J., J.I. Kunzelman, W.J. Kenworthy, and K.K. Hammerstrom. 2003. Depth-related variability in the photobiology of two populations of Halophila johnsonii and Halophila decipiens. Marine Biology 142:1219-1228.

Duronslet, M. J., and coauthors. 1986. The Effects of an Underwater Explosion on the Sea TurtlesLepidochelys kempi and Caretta caretta with Observations of Effects on Other Marine Organisms. Unpublished report submitted to National Marine Fisheries Service Biological Laboratory, Galveston, Texas.

Dustan, P. 1985. Community structure of reef-building corals in the Florida Keys: Carysfort Reef, Key Largo and Long Key Reef, Dry Tortugas. Atoll Research Bulletin 288: 1-27.

Dustan, P. 1977. Vitality of reef coral populations off Key Largo, Florida: recruitment and mortality. Environmental Geology 2(1):51-58.

Dustan, P., and J. C. Halas. 1987. Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. Coral Reefs 6(2):91-106.

Dustan, P., Halas, J.C. 1987. Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. Coral Reefs 6: 91-106.

Eakin, C. M., J. S. Feingold, and P. W. Glynn. 1994. Oil refinery impacts on coral reef communities in Aruba, N.A. Pages 139-145 *in* R. N. Ginsberg, editor. Proceedings of the Colloquium on Global Aspects of Coral Reefs: Health, Hazards and History, 1993. University of Miami, FL.

Edmunds, P. J., and R. Elahi. 2007. The demographics of a 15-year decline in cover of the Caribbean reef coral *Montastraea annularis*. Ecological Monographs 77(1):3-18.

Eiseman, N. and C. McMillan. 1980. A new species of seagrass, Halophila johnsonii, from the Atlantic coast of Florida. Aquatic Botany 9:15-19.

Epstein, N., R. P. M. Bak, and B. Rinkevich. 2000. Toxicity of third generation dispersants and dispersed Egyptian crude oil on Red Sea coral larvae. Marine Pollution Bulletin 40(6):497-503.

Fabricius, K. E., C. Wild, E. Wolanski, and D. Abele. 2003. Effects of transparent exopolymer particles and muddy terrigenous sediments on the survival of hard coral recruits. Estuarine, Coastal and Shelf Science 57(4):613-621.

Fabricius, K., K. Okaji, and G. De'ath. 2010. Three lines of evidence to link outbreaks of the crown-of-thorns seastar Acanthaster planci to the release of larval food limitation. Coral Reefs 29(3):593-605.

Fabry, V. J. 2008. Marine calcifiers in a high-CO_2 ocean. Science 320:1020-1022.

Finney, J. C., and coauthors. 2010. The relative significance of host-habitat, depth, and geography on the ecology, endemism, and speciation of coral endosymbionts in the genus *Symbiodinium*. Microbial Ecology 60(1):250-263.

Fisk, D. A., and V. J. Harriott. 1990. Spatial and temporal variation in coral recruitment on the Great Barrier Reef: Implications for dispersal hypotheses. Marine Biology 107(3):485-490.

Fitt, W. K., and M. E. Warner. 1995. Bleaching patterns of four species of Caribbean reef corals. Biological Bulletin 189(3):298-307.

Flynn, A., S. Rotmann, and C. Sigere. 2006. Long-term monitoring of coral reefs subject to sediment stress in Papua New Guinea. Pages 286-292 *in* Proceedings of the 10th International Coral Reef Symposium, Okinawa, Japan.

Fonseca, M.S., W.J. Kenworthy, and P.E. Whitfield. 2000. Temporal dynamics of seagrass landscapes: a preliminary comparison of chronic and extreme disturbance events. Proc. Fourth International Seagrass Biology Workshop. Biología Marina Mediterranea 7:373-376.

Foster, N. L., I. B. Baums, and P. J. Mumby. 2007. Sexual vs. asexual reproduction in an ecosystem engineer: the massive coral *Montastraea annularis*. Journal of Animal Ecology 76(2):384-391.

Fukami, H., and coauthors. 2004. Geographic differences in species boundaries among members of the *Montastraea annularis* complex based on molecular and morphological markers. Evolution 58(2):324-337.

FWCC. 2000. Benthic Habitat of the Florida Keys. Technical Report TR-4. Florida Fish and Wildlife Conservation Commission. ISSN 1092-194X.

Gallegos, C.L. and W.J. Kenworthy. 1996. Seagrass depth limits in the Indian River Lagoon (Florida, USA): Application of the optical water quality model. Estuarine, Coastal, and Shelf Science 42:267-288.

Garcia, R. P. U., E. M. C. Alvarado, and M. A. Acosta. 1996. Growth of the coral Acropora palmata (Lamarck, 1886) in the Corales del Rosario National Natural Park, Colombian Caribbean. Boletin de Investigaciones Marinas y Costeras 25:7-18.

Garcia-Sais, J. R. 2010. Reef habitats and associated sessile-benthic and fish assemblages across a euphotic–mesophotic depth gradient in Isla Desecheo, Puerto Rico. Coral Reefs 29(2):277-288.

Garzon-Ferreira, J., D. L. Gil-Agudelo, L. M. Barrios, and S. Zea. 2001. Stony coral diseases observed in southwestern Caribbean reefs. Hydrobiologia 460(1):65-69.

Geister, J. 1977. The influence of wave exposure on the ecological zonation of Caribbean coral reefs. Proceedings of the 3rd International Coral Reef Symposium 1: 23-29.

Ghiold, J., and S. H. Smith. 1990. Bleaching and recovery of deep-water, reef-dwelling invertebrates in the Cayman Islands, B.W.I. Caribbean Journal of Science 26(1-2):52-61.

Gilmore, M. D., and B. R. Hall. 1976. Life history, growth habits, and constructional roles of Acropora cervicornis in the patch reef environment. Journal of Sedimentary Research 46(3):519-522.

Gilmour, J. P. 2002. Acute sedimentation causes size-specific mortality and asexual budding in the mushroom coral, Fungia fungites. Marine and Freshwater Research 53(4):805-812.

Ginsburg, R. N., and J. C. Lang, editors. 2003. Status of coral reefs in the western Atlantic: Results of initial surveys, Atlantic and Gulf Rapid Reef Assessment(AGRRA) program, volume 496.

Gitschlag, G. R. 1990. Sea turtle monitoring at offshore oil and gas platforms. Pages 223–246 in 10th Annual Workshop on Sea Turtle Biology and Conservation.

Gitschlag, G. R., and B. A. Herczeg. 1994. Sea turtle observations at explosive removals of energy structures. Mar. Fish. Rev. 56(2):1-8.Gladfelter, E. H., R. K. Monahan, and W. B. Gladfelter. 1978. Growth rates of five reef-building corals in the northeastern Caribbean. Bulletin of Marine Science 28(4):728-734.

Gladfelter, E.H. 1982. Skeletal development in *Acropora cervicornis*: I. Patterns of calcium carbonate accretion in the axile corallite. Coral Reefs 1: 45-52.

Gleason, D. F., B. S. Danilowicz, and C. J. Nolan. 2009. Reef waters stimulate substratum exploration in planulae from brooding Caribbean corals. Coral Reefs 28(2):549-554.

Glynn, P.W. 1990. Feeding ecology of selected coral-reef macroconsumers: Patterns and effects of coral community structure. *In*: Z. Dubinsky (ed.) Ecosystems of the world coral reefs. Elsevier Science Publishers, NY pp.439-452.

Golbuu, Y., S. Victor, E. Wolanski, and R. H. Richmond. 2003. Trapping of fine sediment in a semi-enclosed bay, Palau, Micronesia. Estuarine, Coastal and Shelf Science 57(5-6):941-949.

Goldberg, W. M. 1973. The ecology of the coral octocoral communities off the southeast Florida coast: geomorphology, species composition and zonation. Bulletin of Marine Science 23:465-488.

Goreau, N. I., T. J. Goreau, and R. L. Hayes. 1981. Settling, survivorship and spatial aggregation in planulae and juveniles of the coral *Porites porites* (Pallas). Bulletin of Marine Science 31(2):424-435.

Goreau, T. F. 1959. The ecology of Jamaican coral reefs I. Species composition and zonation. Ecology 40(1):67-90.

Goreau, T. F., and J. W. Wells. 1967. The shallow-water Scleractinia of Jamaica: Revised List of Species and their Vertical Distribution Range. Bulletin of Marine Science 17(2):442-453.

Goreau, T.F., Goreau, N.I. 1973. Coral Reef Project--Papers in Memory of Dr. Thomas F. Goreau. Bulletin of Marine Science 23: 399-464.

Goreau, T.J., Cervino, J.M., Pollina, R. 2004. Increased zooxanthellae numbers and mitotic index in electrically stimulated corals. Symbiosis 37: 107-120.

Graham, E. M., A. H. Baird, and S. R. Connolly. 2008. Survival dynamics of scleractinian coral larvae and implications for dispersal. Coral Reefs 27(3):529-539.

Greening, H. and N. Holland. 2003. Johnson's seagrass (Halophila johnsonii) monitoring workshop. Florida Marine Research Institute, St. Petersburg, Florida. 14 pp.

Gregg, K., Walker, B., Karazsia, J. 2013. Review of Severe Impacts to Coral Reef and Hardbottom in the Federal Channel that Would Result from Expansion of Port Everglades.

Grober-Dunsmore, R., V. Bonito, and T. K. Frazer. 2006. Potential inhibitors to recovery of Acropora palmata populations in St. John, U.S. Virgin Islands. Marine Ecology Progress Series 321:123-132.

Grober-Dunsmore, R., V. Bonito, and T. K. Frazer. 2007. Discernment of sexual recruits is not critical for assessing population recovery of Acropora palmata. Marine Ecology Progress Series 335:233-236.

Hall, L.M., M.D. Hanisak, and R.W. Virnstein. 2006. Fragments of the seagrasses Halodule wrightii and Halophila johnsonii as potential recruits in Indian River Lagoon, Florida. Marine Ecology Progress Series 310:109-117.

Hall, V.R. 1997. Interspecific differences in the regeneration of artificial injuries on scleractinian corals. Journal of Experimental Marine Biology and Ecology, 212:9-23.

Hall, V.R. and T.P. Hughes. 1996. Reproductive strategies of modular organisms: comparative studies of reef-building corals. Ecology, 77:950-963.

Halley, R. B., C. T. Reich, and T. D. Hickey. 2001. Coral reefs in Honduras: Status after Hurricane Mitch, USGS Open File Report 01-133.

Hammerstrom, K. and W. J. Kenworthy. 2002. Investigating the existence of a Halophila johnsonii sediment seed bank. Center for Coastal Fisheries and Habitat Research. NCCOS, NOS, NOAA, Beaufort, North Carolina.

Harrington, L., K. Fabricius, G. De'ath, and A. Negri. 2004. Recognition and selection of settlement substrata determine post-settlement survival in corals. Ecology 85(12):3428-3437.

Harriott, V. J. 1985. Mortality rates of scleractinian corals before and during a mass bleaching event. Marine Ecology Progress Series 21(1):81-88.

Harvell, C. D., and coauthors. 1999. Emerging marine diseases--climate links and anthropogenic factors. Science 285(5433):1505.

Heidelbaugh, W.S., L.M. Hall, W.J. Kenworthy, P.E. Whitfield, R.W. Virnstein, L.J. Morris, and M.D. Hanisak. 2000. Reciprocal transplanting of the threatened seagrass Halophila johnsonii (Johnson's seagrass) in the Indian River Lagoon, Florida. In: Bortone, S.A. (Eds.), Seagrasses: Monitoring, Ecology, Physiology, and Management. CRC Press. Boca Raton, Florida. pp. 197-210.

Helmle, K. P., R. E. Dodge, P. K. Swart, D. K. Gledhill, and C. M. Eakin. 2011. Growth rates of Florida corals from 1937 to 1996 and their response to climate change. Nature Communications 2:215.

Hernández-Pacheco, R., E. A. Hernández-Delgado, and A. M. Sabat. 2011. Demographics of bleaching in a major Caribbean reef-building coral: Montastraea annularis. Ecosphere 2(1):art 9.

Heyward, A.J. and J.D.Collins. 1985. Fragmentation in *Montipora ramosa*: the genet and the ramet concept applied to a coral reef. Coral Reefs, 4:35-40.

Hickerson, E. L., G. P. Schmahl, M. Robbart, W. F. Precht, and C. Caldow. 2008. The state of coral reef ecosystems of the Flower Garden Banks, Stetson Bank, and other banks in the northwestern Gulf of Mexico. Pages 189–217 *in* J. E. Waddell, and A. M. Clarke, editors. The state of coral reef ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA/National Centers for Coastal Ocean Science, Silver Spring, MD.

Highsmith, R. C. 1981. Coral bioerosion: damage relative to skeletal density. American Naturalist 117(2):193-198.

Highsmith, R.C. 1982. Reproduction by fragmentation in corals. Marine Ecology Progress Ser., 7:207-226.

Hill, M. S. 1998. Spongivory on Caribbean reefs releases corals from competition with sponges. Oecologia 117(1-2):143-150.

Hodel, E., and B. Vargas-Angel. 2007. Histopathological assessment and comparison of sedimentation and phosphate stress in the Caribbean staghorn coral, Acropora cervicornis. Microscopy and Microanalysis 13(S02):220-221.

Hoke, S. M. 2007. Gametogenesis and spawning of the elliptical star coral, *Dichocoenia stokesi* (Cnidaria: Scleractinia) in Southeast Florida. Masters' Thesis, Nova Southeastern University, Ft Lauderdale, FL.

Hollarsmith, J. A., S. P. Griffin, and T. D. Moore. 2012. Success of outplanted Acropora cervicornis colonies in reef restoration. 12th International Coral Reef Symposium.Hubbard, D. K., and D. Scaturo. 1985. Growth rates of seven species of scleractinean corals from Cane Bay and Salt River, St. Croix, USVI. Bulletin of Marine Science 36(2):325-338.

Hubbard, J., and Y. Pocock. 1972. Sediment rejection by recent scleractinian corals: a key to palaeo-environmental reconstruction. Geologische Rundschau 61(2):598-626.

Hudson, J. H., and W. B. Goodwin. 1997. Restoration and growth rate of hurricane damaged pillar coral (*Dendrogyra cylindrus*) in the Key Largo National Marine Sanctuary, Florida. Pages 567-570 *in* Proceedings of the 8th International Coral Reef Symposium, Panama City, Panama.

Hughes, T. P. 1985. Life histories and population dynamics of early successional corals. Pages 101-106 *in* C. Gabrie, and B. Salvat editors. Proceedings of the 5th International Coral Reef Congress, Tahiti, French Polynesia.

Hughes, T. P. 1987. Skeletal Density and Growth Form of Corals. Marine Ecology Progress Series 35(3):259-266.

Hughes, T. P. 1994. Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. Science 265(5178):1547-51.

Hughes, T. P. 1996. Demographic approaches to community dynamics: A coral reef example. Ecology 77(7):2256-2260.

Hughes, T. P., and J. B. C. Jackson. 1985. Population-Dynamics and Life Histories of Foliaceous Corals. Ecological Monographs 55(2):141-166.

Hughes, T. P., and J. E. Tanner. 2000. Recruitment failure, life histories, and long-term decline of Caribbean corals. Ecology 81(8):2250-2263.

Hughes, T.P. and J.H. Connell. 1987. Population dynamics based on size or age: a coral reef analysis. American Naturalist, 129:818-829.

Hughes, T.P., D. Ayre, and J.H. Connell. 1992. The evolutionary ecology of corals. J. Ecol. Evol., 7:292-295.

Humann, P. 1993. Reef coral identification: Florida, Caribbean, Bahamas. New World Publications, Inc., Jacksonville, FL.

Humphrey, C., M. Weber, C. Lott, T. Cooper, and K. Fabricius. 2008. Effects of suspended sediments, dissolved inorganic nutrients and salinity on fertilisation and embryo development in the coral Acropora millepora (Ehrenberg, 1834). Coral Reefs 27(4):837-850.

Hunter, I. G., and B. Jones. 1996. Coral associations of the Pleistocene Ironshore Formation, Grand Cayman. Coral Reefs 15(4):249-267.

Huntington, B. E., M. Karnauskas, and D. Lirman. 2011. Corals fail to recover at a Caribbean marine reserve despite ten years of reserve designation. Coral Reefs 30(4):1077-1085.

Idjadi, J. A., and coauthors. 2006. Rapid phase-shift reversal on a Jamaican coral reef. Coral Reefs 25(2):209-211.

IPCC. 2013. Working Group I Contribution to the IPCC Fifth Assessment Report (AR5), Climate Change 2013: The Physical Science Basis. Technical Summary - Final Draft Underlying Scientific-Technical Assessment, Stockholm.

IUCN. 2010. IUCN Red List of Threatened Species. Version 3.1. Page ii + 30 pp. Page: http://www.iucnredlist.org/. IUCN Species Survival Commission, Gland, Switzerland and Cambridge, UK.

IUCN. 2013. Status and trends of Caribbean coral reefs: 1969-2012. Global Coral Reef Monitoring Network, Washington, D. C.

Jaap, W. C. 1974. Scleractinian growth rate studies.

Jaap, W. C. 1984. The ecology of south Florida coral reefs: A community profile, FWS/OBS-82/08.

Jaap, W. C. 1985. An epidemic zooxanthellae expulsion during 1983 in the lower coral reefs: hyperthermic etiology. Pages 143-148 *in* Proceedings of The Fifth International Coral Reef Congress, Tahiti, Polynesia.

Jaap, W. C., W. G. Lyons, P. Dustan, and J. C. Halas. 1989. Stony coral (Scleractinia and Milleporina) community structure at Bird Key Reef, Ft. Jefferson National Monument, Dry Tortugas, Florida.

Jaap, W.C. 1974. Scleractinian growth rate studies. Proceedings of the Florida Keys Coral Reef Workshop. Florida Department of Natural Resources Coastal Coordinating Council p 17.

Jaap, W.C. 1979. Observation on zooxanthellae expulsion at Middle Sambo Reef, Florida Keys. Bulletin of Marine Science 29: 414-422.

Jaap, W.C. 1984. The ecology of the south Florida coral reefs: a community profile. US Fish and Wildlife Service (139).

Jaap, W.C., Lyons, W.G., Dustan, P., Halas, J.C. 1989. Stony coral (Scleractinia and Milleporina) community structure at Bird Key Reef, Ft. Jefferson National Monument, Dry Tortugas, Florida. Florida Marine Research Publication 46: 31.

Jackson, J.B.C. 1985. Distribution and ecology of clonal and aclonal benthic invertebrates. *In*: Jackson, J.B.C., Buss, L.W., Cook, R.E. (eds.). Population Biology and Evolution of Clonal Organisms. Yale University Press, New Haven, CT, pp. 297-356.

Jackson, J.B.C. and S.R. Palumbi. 1979. Regeneration and partial predation in cryptic coral reef environments: Preliminary experiments on sponges and ectoprocts. Colloq. Int. C.N.R.S., 291:303-308.

Jewett-Smith, J., C. McMillan, W.J. Kenworthy, and K. Bird. 1997. Flowering and genetic banding patterns of Halophila johnsonii and conspecifics. Aquatic Botany 59 (1997). pp. 323-331.

Jokiel, P. L., and coauthors. 2008. Ocean acidification and calcifying reef organisms: a mesocosm investigation. Coral Reefs 27(3):473-483.

Kahn, A. E. and M.J. Durako. 2008. Photophysiological responses of Halophila johnsonii to experimental hyposaline and hyper-CDOM conditions. Journal of Experimental Marine Biology and Ecology 367:230-235.

Karlson, R.H. 1986. Disturbance, colonial fragmentation, and size-dependent life history variation in two coral reef cnidarians. Mar. Ecol. Prog. Ser., 28:245-249.

Karlson, R.H. 1988. Size-dependent growth in two zoanthid species: a contrast in clonal strategies. Ecology, 69:1219-1232.

Keck, J., R. S. Houston, S. Purkis, and B. M. Riegl. 2005. Unexpectedly high cover of Acropora cervicornis on offshore reefs in Roatán (Honduras). Coral Reefs 24(3):509.

Keller, B. D., J. B. C. Jackson, and (eds). 1991. Long-term assessment of the oil spill at Bahia Las Minas, Panama, interim report, volume I: executive summary. US Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, OCS Study MMS 90-0030, New Orleans, La.

Kendall, J., E. N. Powell, S. J. Connor, and T. J. Bright. 1983. The effects of drilling fluids (muds) and turbidity on the growth and metabolic state of the coral Acropora cervicornis, with comments on methods of normalization for coral data. Bulletin of Marine Science 33(2):336-352.

Kenworthy, W.J. 1993. The distribution, abundance, and ecology of Halophila johnsonii (Eiseman) in the lower Indian River, Florida. Final Report to NMFS Office of Protected Resources. 72 pp.

Kenworthy, W.J. 1997. An updated biological status review and summary of the Proceedings of a Workshop to Review the Biological Status of the Seagrass, Halophila johnsonii (Eiseman). Report submitted to NMFS Office of Protected Resources, October 15, 1997. 24 pp.

Kenworthy, W.J. 1999. Demography, population dynamics, and genetic variability of natural and transplanted populations of Halophila johnsonii, a threatened seagrass. Annual Progress Report, July 1999.

Kenworthy, W.J. 2000. The role of sexual reproduction in maintaining populations of Halophila decipiens: implications for the biodiversity and conservation of tropical seagrass ecosystems. Pacific Conservation Biology 5:260-268.

Kenworthy, W.J. M.S. Fonseca, P.E. Whitfield, and K. Hammerstrom. 2002. Analysis of seagrass recovery in experimental excavations and propeller-scar disturbances in the Florida Keys National Marine Sanctuary. Journal of Coastal Research 37:75-85.

Kenworthy, W.J. and D.E. Haunert. 1991. The light requirements of seagrasses: proceedings of a workshop to examine the capability of water quality criteria, standards, and monitoring programs to protect seagrasses. NOAA Technical Memorandum NMFS-SEFSC-287.

Kenworthy, W.J. and M.S. Fonseca. 1996. Light requirements of seagrasses Halodule wrightii and Syringodium filiforme derived from the relationship between diffuse light attenuation and maximum depth distribution. Estuaries 19:740-750.

Kenworthy, W.J., S. Wyllie-Echeverria, R.G. Coles, G. Pergent, and C. Pergent-Martini. 2006. Seagrass Conservation Biology: An Interdisciplinary Science for Protection of the Seagrass Biome. In: Larkum, A.W.D., Orth, R.J., Duarte, C.M., (Eds.), Seagrasses: Biology, Ecology, and Conservation. Springer. Dordrecht, The Netherlands. pp. 595-623.

Ketten, D., and S. M. Bartol. 2005. Functional Measures of Sea Turtle Hearing. Final Report. Prepared by Woods Hole Oceanographic Institution, Woods Hole, MA for Office of Naval Research.

Kinzie, R. A., 3rd, M. Takayama, S. R. Santos, and M. A. Coffroth. 2001. The adaptive bleaching hypothesis: experimental tests of critical assumptions. Biol Bull 200(1):51-8.

Kinzie, R.A. III. 1973. The zonation of west-Indian gorgonians. Bulletin of Marine Science, 23:93-155.

Kirsch, K.D., K.A. Barry, M.S. Fonseca, P.E. Whitfield, S.R. Meehan, W.J. Kenworthy, and B.E. Julius. 2005. The Mini-312 Program - an expedited damage assessment and restoration process for seagrasses in the Florida Keys National Marine Sanctuary. Journal of Coastal Research SI40:109-119.

Klima, E. F., G. R. Gitschlag, and M. L. Renard. 1988. Impacts of the explosive removal of offshore petroleum platforms on sea turtles and dolphins. Marine Fisheries Review 50(3):33-42.

Knutson, S., C. A. Downs, and R. H. Richmond. 2012. Concentrations of Irgarol in selected marinas of Oahu, Hawaii and efects on settlement of coral larvae. Ecotoxicology 21:1-8.

Kobayashi, A. 1984. Regeneration and regrowth of fragmented colonies of hermatypic corals *Acropora formosa* and *Acropora nasuta*. Galaxea, 291:13-23.

Kojis, B.L. and N.J. Quinn. 1985. Puberty in *Goniastrea favulus* age or size limited? Proc. 5th Int. Coral Reef Symp (Tahiti), 4:289-293.

Kornicker, L.S. and D.W. Boyd. 1962. Shallow-water geology and environments of Alacrán Reef complex, Campeche Bank, Mexico. Bulletin of the American Association of Petroleum Geologists, 46:640-673.

Kramer, P. R. 2002. Status and Trends Working Group Report. Pages 28-37 *in* A. W. Bruckner, editor Proceedings of the Caribbean Acropora workshop: Potential application of the U.S. Endangered Species Act as a conservation strategy. NOAA Technical Memorandum NMFS-OPR-24, Silver Spring, MD.

Krediet, C. J., and coauthors. 2009. Utilization of Mucus from the Coral *Acropora palmata* by the Pathogen *Serratia marcescens* and by Environmental and Coral Commensal Bacteria. Antimicrobial Agents & Chemotherapy 75(12):3851-3858.

Kuffner, I. B., A. J. Andersson, P. L. Jokiel, K. S. Rodgers, and F. T. Mackenzie. 2008. Decreased abundance of crustose coralline algae due to ocean acidification. Nature Geoscience 1(2):114-117

Kuntz, N. M., D. I. Kline, S. A. Sandin, and F. Rohwer. 2005. Pathologies and mortality rates caused by organic carbon and nutrient stressors in three Caribbean coral species. Marine Ecology Progress Series 294:173-180.

Kunzelman, J.I. 2007. Southern Range, permanent transect implementation, summer sampling 2006. Report prepared for the Johnson's Seagrass Recovery Team. Florida Fish and Wildlife Conservation Commission, St. Petersburg, Florida. 23 pp.

LaJeunesse, T. 2002. Diversity and community structure of symbiotic dinoflagellates from Caribbean coral reefs. Marine Biology 141(2):387-400.

Landry, J. B., W. J. Kenworthy, and G. Di Carlo. 2008. The effects of docks on seagrasses, with particular emphasis on the threatened seagrass, Halophila johnsonii. Report submitted to NMFS Office of Protected Resources, July 2008. 31 pp.

Langdon, C., and coauthors. 2003. Effect of elevated CO_2 on the community metabolism of an experimental coral reef. Global Biogeochemical Cycles 17(1):1-14.

Lapointe, B. E., P. J. Barile, M. M. Littler, and D. S. Littler. 2005. Macroalgal blooms on southeast Florida coral reefs II. Cross-shelf discrimination of nitrogen sources indicates widespread assimilation of sewage nitrogen. Harmful Algae 4:1106-1122.

Lasker, H. R. 1980. Sediment rejection by reef corals: the roles of behavior and morphology in Montastrea cavernosa (Linnaeus). Journal of Experimental Marine Biology and Ecology 47(1):77-87.

Lasker, H.R. 1990. Clonal propagation and population dynamics of a gorgonian coral. Ecology, 7:1578-1589.

Leder, J. J., A. M. Szmant, and P. K. Swart. 1991. The effect of prolonged "bleaching" on skeletal banding and stable isotopic composition in *Montastrea annularis*. Coral Reefs 10(1):19-27.

Lendhardt, M. L. 1994. Seismic and very low frequency sound induced behaviors in captive loggerhead marine turtles (Caretta caretta). In Proceedings of the fourteenth annual symposium on sea turtle biology and conservation (K.A. Bjorndal, A.B. Bolten, D.A. Johnson & P.J. Eliazar, eds.) NOAA Technical Memorandum, NMFS-SEFC-351, National Technical Information Service, Springfield, Virginia, 238-241.

Levitan, D. R., and coauthors. 2004. Mechanisms of reproductive isolation among sympatric broadcast-spawning corals of the *Montastraea annularis* species complex. Evolution 58(2):308-323.

Lewis, C, S.L. Slade, K.E. Maxwell, and T. Matthews. 2009. Lobster Trap Impact on Coral Reefs: Effects of wind-driven trap movement. New Zealand Journal of Marine and Fresh Water Fisheries. 43: 271-282.

Lewis, J. B. (1974). The settlement behaviour of planulae larvae of the hermatypic coral Favia fragum (Esper). J. exp. mar. Biol. Ecol. 15: 165-172.

Lewis, J. B. 1974. The settlement behaviour of planulae larvae of the hermatypic coral *Favia fragum* (Esper). Journal of Experimental Marine Biology and Ecology 15:165-172.

Lewis, J.B. 1977. Suspension feeding in Atlantic reef corals and the importance of suspended particulate matter as a food source. Proceedings of the 3rd International Coral Reef Symposium 1: 405-408.

Lewison, R. L., S. A. Freeman and L. B. Crowder (2004). "Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles." Ecology Letters 7: 221-231.

Lidz, B. H., and D. G. Zawada. 2013. Possible Return of Acropora cervicornis at Pulaski Shoal, Dry Tortugas National Park, Florida. Journal of Coastal Research 29(2):256-271.

Lighty, R.G., I.G. Macintyre, and R. Stuckenrath. 1982. *Acropora palmata* reef framework: A reliable indicator of sea level in the western Atlantic for the past 10,000 years. Coral Reefs, 1:125-130.

Lirman, D. 2000. Lesion regeneration in the branching coral Acropora palmate: effects of colonization, colony size, lesion size, and lesion shape. Marine Ecology Progress Series 197: 209-215.

Lirman, D., and coauthors. 2010. A window to the past: documenting the status of one of the last remaining 'megapopulations' of the threatened staghorn coral Acropora cervicornis in the Dominican Republic. Aquatic Conservation: Marine and Freshwater Ecosystems 20(7):773-781.

Lopez, J. V., R. Kersanach, S. A. Rehner, and N. Knowlton. 1999. Molecular determination of species boundaries in corals: Genetic analysis of the Montastraea annularis complex using amplified fragment length polymorphisms and a microsatellite marker. Biological Bulletin 196(1):80-93.

Loya, Y. 1976. Recolonization of Red Sea Corals Affected by Natural Catastrophes and Man-Made Perturbations. Ecology 57: 278–289.

Loya, Y., and B. Rinkevich. 1980. Effects of oil pollution on coral reef communities. Marine Ecology Progress Series 3(16):167-180.

Lundgren, I., and Z. Hillis-Starr. 2008. Variation in *Acropora palmata* bleaching across benthic zones at Buck Island Reef National Monument (St. Croix, USVI) during the 2005 thermal stress event. Bulletin of Marine Science 83:441-451.

Lunz, K. S. 2013. Final Report Permit Number: FKNMS-2010-126-A3. Florida Fish and Wildlife Conservation Commission, St. Petersburg, FL.

Marszalek, D. S. 1981. Impact of dredging on a subtropical reef community, Southeast Florida, USA. Pages 147-153 *in* Proceedings of the Fourth International Coral Reef Symposium, Manila, volume 1.

Marubini, F., and P. S. Davies. 1996. Nitrate increases zooxanthellae population density and reduces skeletogenesis in corals. Marine Biology 127(2):319-328.

Mayor, P. A., C. S. Rogers, and Z. M. Hillis-Starr. 2006. Distribution and abundance of elkhorn coral, Acropora palmata, and prevalence of white-band disease at Buck Island Reef National Monument, St. Croix, US Virgin Islands. Coral Reefs 25(2):239-242.

McCauley, R. D., and coauthors. 2000a. Marine seismic surveys: a study of environmental implications. APPEA Journal 40:692-708.

McCauley, R. D., and coauthors. 2000b. Marine Seismic Surveys: Analysis and Propagation of Air-gun Signals; and Effects of Air-gun Exposure on Humpback Whales, Sea Turtles, Fishes and Squid. Centre for Marine Science and Technology, Western Australia.

McClanahan, T. R., and N. A. Muthiga. 1998. An ecological shift in a remote coral atoll of Belize over 25 years. Environmental Conservation 25(2):122-130.

McField, M. D. 1999. Coral response during and after mass bleaching in Belize. Bulletin of Marine Science 64(1):155-172.

Meester, E.H., M. Noordeloos, and R.P.M. Bak. 1994. Damage and regeneration: links to growth in the reef building coral *Montastrea annularis*. Mar. Ecol. Proc. Ser., 121:119-128.

Mendes, J. M., and J. D. Woodley. 2002. Effect of the 1995-1996 bleaching event on polyp tissue depth, growth, reproduction and skeletal band formation in Montastraea annularis. Marine Ecology Progress Series 235:93-102.

Miller, J., and coauthors. 2009. Coral disease following massive bleaching in 2005 causes 60% decline in coral cover on reefs in the US Virgin Islands. Coral Reefs 28(4):925-937.

Miller, M. W., and D. E. Williams. 2007. Coral disease outbreak at Navassa, a remote Caribbean island. Coral Reefs 26(1):97-101.

Miller, S. L., M. Chiappone, L. M. Rutten, and D. W. Swanson. 2008. Population status of Acropora corals in the Florida Keys. Proceedings of the 11th International Coral Reef Symposium:775-779.

Miller, S. L., W. F. Precht, L. M. Rutten, and M. Chiappone. 2013. Florida Keys Population Abundance Estimates for Nine Coral Species Proposed for Listing Under the U.S. Endangered Species Act., 1(1), Dania Beach, Florida.

Miller, S.L., M. Chiappone, and L.M. Rutten. 2007. 2007-Quick look report: Large-scale assessment of *Acropora* corals, coral species richness, urchins and *Coralliophila* snails in the Florida Keys National Marine Sanctuary and Biscayne National Park. Center for Marine Science, University of North Carolina-Wilmington, Key Largo, Florida. 147 pp.

Miller, S.L., M. Chiappone, L.M. Rutten, and D.W. Swanson. 2008. Population status of *Acropora* corals in the Florida Keys. Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, Florida, July 7-11. Session Number 18.

Milliken, T. and H. Tokunaga (1987). The Japanese sea turtle trade 1970-1986. A special report prepared by TRAFFIC (Japan). Washington, D.C. , Center for Environmental Education**:** 171.

Moein, S. E., and coauthors. 1994. Evaluation of seismic sources for repelling sea trutles from hopper dredges. In Sea Turtle Research Program, Summary Report. Final Report. Prepared for US Army Engineer Division, South Atlantic, Atlanta, GA, and US Naval Submarine Base, Kings Bay, GA. Technical Report CERC-95. Original not seen, cited in Moein-Bartol S.E. 2008. Review of auditory function of sea turtles. Bioacoustics 2008: 57-59. Viewed online March 2011 at http://www.seaturtle.org/PDF/BartolSM_2008_Bioacoustics.pdf.

Morelock, J., W. R. Ramírez, A. W. Bruckner, and M. Carlo. 2001. Status of coral reefs, southwest Puerto Rico. Caribean Journal of Science, special publication 4:57.

Morse, A. N. C., and coauthors. 1996. An ancient chemosensory mechanism brings new life to coral reefs. Biological Bulletin 191:149-154.

Morse, A.N.C., and Morse, D.E. 1996. Flypapers for coral and other planktonic larvae. BioScience 46: 254-262.

Morse, D. E., A. N. C. Morse, P. T. Raimondi, and N. Hooker. 1994. Morphogen-based chemical flypaper for *Agaricia humilis* coral larvae. Biological Bulletin 186:172-181.

Morse, D. E., N. Hooker, A. N. C. Morse, and R. A. Jensen. 1988. Control of larval metamorphosis and recruitment in sympatric agariciid corals. Journal of Experimental Marine Biology and Ecology 116(3):193-217.

Muller, E. M., C. S. Rogers, A. S. Spitzack, and R. van Woesik. 2008. Bleaching increases likelihood of disease on Acropora palmata (Lamarck) in Hawksnest Bay, StJohn, US Virgin Islands. Coral Reefs 27(1):191-195.

Muller, E. M., C. S. Rogers, and R. Woesik. 2013. Early signs of recovery of Acropora palmata in St. John, US Virgin Islands. Marine Biology.

Mumby, P. J., A. Hastings, and H. J. Edwards. 2007. Thresholds and the resilience of Caribbean coral reefs. Nature 450(7166):98-101.

Muscatine, L., D. Grossman, and J. Doino. 1991. Release of symbiotic algae by tropical sea anemones and corals after cold shock. Marine Ecology Progress Series 77(2):233-243.

National Marine Fisheries Service. 2007. Endangered Species Act 5-Year Review: Johnson's Seagrass (Halophila johnsonii, Eiseman). Prepared by NOAA's Johnson's Seagrass Status Review Team, Silver Spring, Maryland. 58 pp.

Negri, A. P., and A. J. Heyward. 2000. Inhibition of fertilization and larval metamorphosis of the coral *Acropora millepora* (Ehrenberg, 1834) by petroleum products. Marine Pollution Bulletin 41(7-12):420-427.

Negri, A. P., and A. J. Heyward. 2001. Inhibition of coral fertilisation and larval metamorphosis by tributyltin and copper. Marine Environmental Research 51(1):17-27.

Negri, A. P., N. S. Webster, R. T. Hill, and A. J. Heyward. 2001. Metamorphosis of broadcast spawning corals in response to bacterial isolated from crustose algae. Marine Ecology Progress Series 223:121-131.

Negri, A. P., P. A. Marshall, and A. J. Heyward. 2007. Differing effects of thermal stress on coral fertilization and early embryogenesis in four Indo Pacific species. Coral Reefs 26(4):759-763.

Neigel, J.E. and J.C. Avise. 1983. Clonal diversity and population structure in a reef-building coral, *Acropora cervicornis*: self-recognition analysis and demographic interpretation. Evolution, 37:437-454.

Nugues, M. M. 2002. Impact of a coral disease outbreak on coral communities in St. Lucia: What and how much has been lost? Marine Ecology Progress Series 229:61-71.

O'Hara, J., and J. R. Wilcox. 1990. Avoidance Responses of Loggerhead Turtles, Caretta caretta, to Low Frequency Sound. Copeia 1990(2):564-567.

Oppenheimer, C.H., 1963. Effects of Hurricane Carla on the ecology of Redfish Bay, Texas. Bulletin Marine Science Gulf Caribbean 13:59-72.

Oxenford, H. A., and coauthors. 2008. Quantitative observations of a major coral bleaching event in Barbados, Southeastern Caribbean. Climatic Change 87(3-4):435-449.

Pait, A. S., and coauthors. 2007. An assessment of chemical contaminants in the marine sediments of southwest Puerto Rico, Silver Spring, MD.

Patterson, K. L., and coauthors. 2002. The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, *Acropora palmata*. Proceedings of the National Academy of Sciences 99(13):8725-8730.

Philbrick, C. T. and D.H. Les. 1996. Evolution of aquatic angiosperm reproductive systems. BioScience 46:813-826.

Philipp, E., and K. Fabricius. 2003. Photophysiological stress in scleractinian corals in response to short-term sedimentation. Journal of Experimental Marine Biology and Ecology 287(57-78).

Porter, J. W. 1976. Autotrophy, heterotrophy, and resource partitioning in Caribbean reef-building corals. The American Naturalist 110(975):731-742.

Porter, J. W. 1987. Species profiles: 1ife histories and environmental requirements o f coastal fishes and invertebrates (south Florida) --reef-building corals. U.S. Fish Wildl . Serv. Biol. Rep. 82(11.73). U.S. Army Corps of Engineers, TR EL-82-4. 23 pp.

Porter, J. W. 1987. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates (south Florida): reef-building corals.

Porter, J. W., W. K. Fitt, H. J. Spero, C. S. Rogers, and M. W. White. 1989. Bleaching in reef corals: physiological and stable isotopic responses. Proceedings of the National Academy of Sciences of the United States of America 86:9342-9346.

Porter, J.W. 1976. Autotrophy, heterotrophy, and resource partitioning in Caribbean reef corals. American Naturalist, 110:731-742.

Precht, W., R. Aronson, R. Moody, and L. Kaufman. 2010. Changing patterns of microhabitat utilization by the threespot damselfish, *Stegastes planifrons*, on Caribbean Reefs. PLoS ONE 5(5):171-233.

Rachello-Dolmen, P. G., and D. F. R. Cleary. 2007. Relating coral species traits to environmental conditions in the Jakarta Bay/Pulau Seribu reef system, Indonesia. Estuarine, Coastal and Shelf Science 73(3-4):816-826.

Randall, C. J., and A. M. Szmant. 2009. Elevated temperature affects development, survivorship, and settlement of the elkhorn coral, *Acropora palmata* (Lamarck 1816). Biological Bulletin 217:269-282.

Rasher, D. B., and coauthors. 2012. Effects of herbivory, nutrients, and reef protection on algal proliferation and coral growth on a tropical reef. Oecologia 169(1):187-198.

Rasher, D. B., and M. E. Hay. 2010. Chemically rich seaweeds poison corals when not controlled by herbivores. Proceedings of the National Academy of Sciences of the United States of America 107(21):9683-9688.

Rasher, D. B., E. P. Stout, S. Engel, J. Kubanek, and M. E. Hay. 2011. Macroalgal terpenes function as allelopathic agents against reef corals. Proceedings of the National Academy of Sciences of the United States of America 108(43):17726-17731.

Reichelt-Brushett, A. J., and P. L. Harrison. 2000. The effect of copper on the settlement success of larvae from the scleractinian coral *Acropora tenuis*. Marine Pollution Bulletin 41(7-12):385-391.

Reichelt-Brushett, A. J., and P. L. Harrison. 2005. The effect of selected trace metals on the fertilization success of several scleractinian coral species. Coral Reefs 24(4):524-534.

Renegar, D. A., and B. M. Riegl. 2005. Effect of nutrient enrichment and elevated CO_2 partial pressure on growth rate of Atlantic scleractinian coral *Acropora cervicornis*. Marine Ecology Progress Series 293:69-76.

Richardson, L. L. 1998. Coral diseases: what is really known? Trends in Ecology & Evolution 13(11):438-443.

Richardson, L. L., and J. D. Voss. 2005. Changes in a coral population on reefs of the northern Florida Keys following a coral disease epizootic. Marine Ecology Progress Series 297:147-156.

Richmond RH, Hunter CL (1990) Reproduction and recruitment of corals: comparisons among the Caribbean, the tropical Pacific and the Red Sea. Mar Ecol Prog Ser 60:185–203.

Richmond, R. H., and C. L. Hunter. 1990. Reproduction and recruitment of corals: Comparisons among the Caribbean, the Tropical Pacific, and the Red Sea. Marine Ecology Progress Series 60(1):185-203.

Riegl, B., and G. M. Branch. 1995. Effects of sediment on the energy budgets of four scleractinian (Bourne 1900) and five alcyonacean (Lamouroux 1816) corals. Journal of Experimental Marine Biology and Ecology 186(2):259-275.

Riegl, B., S. J. Purkis, J. Keck, and G. P. Rowlands. 2009. Monitored and modeled coral population dynamics and the refuge concept. Marine Pollution Bulletin 58(1):24-38.

Rinkevich, B. and Y. Loya. 1989. Reproduction in regeneration colonies of the coral *Stylophora pistillata*. *In*: E. Spainer, Y. Steinberger, N. Luria (eds) Environmental quality and ecosystems stability: v. IV-B Environmental Quality. ISSEQS, Israel.

Ritson-Williams, R., V. J. Paul, S. N. Arnold, and R. S. Steneck. 2009. Larval settlement preferences and post-settlement survival of the threatened Caribbean corals *Acropora palmata* and *A. cervicornis*. Coral Reefs.

Ritson-Williams, R., V. J. Paul, S. N. Arnold, and R. S. Steneck. 2010. Larval settlement preferences and post-settlement survival of the threatened Caribbean corals *Acropora palmata* and *A. cervicornis*. Coral Reefs 29(1):71-81.

Rodriguez-Ramirez, A., and coauthors. 2010. Recent dynamics and condition of coral reefs in the Colombian Caribbean. Revista de Biologia Tropical 58:107-131.

Rogers, C. S. 1979. The effect of shading on coral reef structure and function. Journal of Experimental Marine Biology and Ecology 41(3):269-288.

Rogers, C. S. 1983. Sublethal and lethal effects of sediments applied to common Caribbean reef corals in the field. Marine Pollution Bulletin 14(10):378-382.

Rogers, C. S., and E. M. Muller. 2012. Bleaching, disease and recovery in the threatened scleractinian coral *Acropora palmata* in St. John, US Virgin Islands: 2003–2010. Coral Reefs 31(3):807-819.

Rogers, C. S., Fitz, H. C., Gilnack, M., Beets, J., Hardin, J. (1984). Scleractinian coral recruitment patterns at Salt River Submarine Canyon. St. Croix, U.S.V.I. Coral Reefs 3: 69-76.

Rogers, C. S., H. C. I. Fitz, M. Gilnack, J. Beets, and J. Hardin. 1984. Scleractinian coral recruitment patterns at Salt River submarine canyon, St. Croix, U.S. Virgin Islands. Coral Reefs 3:69-76.

Rogers, C. S., T. H. Suchanek, and F. A. Pecora. 1982. Effects of Hurricanes David and Frederic (1979) on shallow *Acropora palmata* reef communities: St. Croix, U.S. Virgin Islands. Bulletin of Marine Science 32(2):532-548.

Rogers, C., and coauthors. 2002. Acropora in the U.S. Virgin Islands: a wake or an awakening? A status report prepared for the National Oceanographic and Atmospheric Administration. Pages 99-122 *in* A. W. Bruckner, editor. Proceedings of the Caribbean Acropora workshop: potential application of the U.S. Endangered Species Act as a conservation strategy. NOAA Technical Memorandum NMFS-OPR-24, Silver Spring, MD.

Rogers, C.S., Suchanek, T., Pecora, F. 1982. Effects of Hurricanes David and Frederic (1979) on shallow *Acropora* palmata reef communities: St. Croix, USVI. Bulletin of Marine Science 32: 532-548.

Ross, C., R. Ritson-Williams, K. Olsen, and V. J. Paul. 2013. Short-term and latent post-settlement effects associated with elevated temperature and oxidative stress on larvae from the coral Porites astreoides. Coral Reefs 32(1):71-79.

Rotjan, R. D., and coauthors. 2006. Chronic parrotfish grazing impedes coral recovery after bleaching. Coral Reefs 25(3):361-368.

Rotjan, R. D., and S. M. Lewis. 2008. Impact of coral predators on tropical reefs. Marine Ecology Progress Series 367:73-91.

Rotjan, R., and S. Lewis. 2006. Parrotfish abundance and selective corallivory on a Belizean coral reef. Journal of Experimental Marine Biology and Ecology 335(2):292-301.

Ruzicka, R. R., and coauthors. 2013. Temporal changes in benthic assemblages on Florida Keys reefs 11 years after the 1997/1998 El Niño. Marine Ecology Progress Series 489:125-141.

Rylaarsdam, K. W. 1983. Life histories and abundance patterns of colonial corals on Jamaican reefs. Marine Ecology Progress Series 13:249-260.

Sammarco, P. W. 1980. *Diadema* and its relationship to coral spat mortality: grazing, competition, and biological disturbance. J. Exp. Mar. Biol. Ecol. 45(2-3):245-272.

Sammarco, P. W. 1985. The Great Barrier Reef vs. the Caribbean; comparisons of grazers, coral recruitment patterns and reef recovery. Proceedings of the 5th international Coral Reef Congress 4:391-397.

Sammarco, P.W. 1980. *Diadema* and its relationship to coral spat mortality: grazing, competition, and biological disturbance. Journal of Experimental Marine Biology and Ecology, 45:245-272.

Sammarco, P.W. 1985. The Great Barrier Reef vs. the Caribbean: Comparisons of grazers,coral recruitment patterns and reef recovery. Proceedings of the 5th International Coral Reef Congress 4: 391-397.

Sargent, F.J., T.J. Leary, D.W. Crewz, and C.R. Kruer. 1995. Scarring of Florida's seagrasses: assessment and management options. FMRI Technical Report TR-1. Florida Marine Research Institute, St. Petersburg, Florida. 37 pp.

Scatterday, J.W. 1974. Reefs and associated coral assemblages off Bonaire, Netherlands Antilles, and their bearing on Pleistocene and Recent reef models. Proceedings of the 2nd International Coral Reef Symposium 2: 85-106.

Schärer, M., and coauthors. 2009. Elkhorn Coral Distribution and Condition throughout the Puerto Rican Archipelago. Proceedings of the 11th International Coral Reef Symposium, Ft. Lauderdale, Florida.

Schnitzler, C. E., L. L. Hollingsworth, D. A. Krupp, and V. M. Weis. 2012. Elevated temperature impairs onset of symbiosis and reduces survivorship in larvae of the Hawaiian coral, Fungia scutaria. Marine Biology 159(3):633-642.

Shafer, D. J. and J. Robinson. 2001. Evaluation of the use of grid platforms to minimize shading impacts to seagrasses. WRAP Technical Notes Collection (ERDCTN-WRAP-01-02), U.S. Army Corps of Engineers, Research and Development Center, Vicksburg, Mississippi.

Shafer, Deborah J., J. Karazsia, L. Carrubba, and C. Martin. 2008. Evaluation of regulatory guidelines to minimize impacts to seagrasses from single-family residential dock structures in Florida and Puerto Rico. Final report, October 2008. 47 pp.

Shearer, T. L., and M. A. Coffroth. 2006. Genetic identification of Caribbean scleractinian coral recruits at the Flower Garden Banks and the Florida Keys. Marine Ecology Progress Series 306:133-142.

Sheppard, C. and R. Rioja-Nieto. 2005. Sea surface temperature 1871-2099 in 38 cells in the Caribbean region. Marine Environmental Research 60:389-396.

Sheridan, P., Hill, R., and Kojis, B. 2006. Trap fishing in the U.S. Virgin Islands: How and where effort is exerted. Proceedings of the Gulf and Caribbean Fisheries Institute 57: 175-188.

Sheridan, P., Hill, R., Matthews, G., Appeldoorn, R., Kojis, B., and Matthews, T. 2005. Does trap fishing impact coral reef ecosystems? An update. Proceedings of the Gulf and Caribbean Fisheries Institute 56: 511-519.

Shinn, E. 1963. Spur and groove formation on the Florida Reef Tract. Journal of Sedimentary Petrology 33(2):291-303.

Shinn, E. A. 1966. Coral growth-rate, and environmental indicator. Journal of Paleontology 40(2):233-240.

Shinn, E. A. 1976. Coral reef recovery in Florida and the Persian Gulf. Environmental Geology 1:241-254.

Shinn, E.A. 1963. Spur-and-groove formation on the Florida Reef Tract. Journal of Sedimentary Petrology 33: 291-303.

Shinn, E.A. 1966. Coral growth-rate: An environmental indicator. Journal of Paleontology 40: 233-240.

Shinn, E.A. 1976. Coral reef recovery in Florida and the Persian Gulf. Environmental Geology 1: 241-254.

Short, F.T. and H.A. Neckles. 1998. The effects of global climate change on seagrasses. Aquatic Botany 63:169-196.

Short, F.T., W.C. Dennison, T.J.B. Carruthers, and M. Waycott. 2007. Global seagrass distribution and diversity: a bioregional model. Journal of Experimental Marine Biology and Ecology 350:3-20.

Smith, S. R., and R. B. Aronson. 2006. Population dynamics of *Montastraea* spp. in the Florida Keys' Fully Protected Zones: modeling future trends.

Smith, S. R., R. B. Aronson, and J. Ogden. 2008. Continuing decline of *Montastraea* populations on protected and unprotected reefs in the Florida Keys National Marine Sanctuary. 11th International Coral Reef Symposium, Ft. Lauderdale, FL.

Smith, T. B., and coauthors. 2010. Benthic structure and cryptic mortality in a Caribbean mesophotic coral reef bank system, the Hind Bank Marine Conservation District, US Virgin Islands. Coral Reefs 29(2):289-308.

Soong, K. 1991. Sexual reproductive patterns of shallow-water reef corals in Panama. Coral Reefs, 49:832–846.

Soong, K. and J.C. Lang. 1992. Reproductive integration in coral reefs. Biol Bull, 183:418-431.

Soong, K., and J. C. Lang. 1992. Reproductive intergration in reef corals. Biological Bulletin 183:418-431.

Species Act. U.S. Dep. Commer.

Stafford-Smith, M. G. 1993. Sediment-rejection efficiency of 22 species of Australian scleractinian corals. Marine Biology 115(2):229-243.

Stafford-Smith, M. G., and R. F. G. Ormond. 1992. Sediment-rejection mechanisms of 42 species of Australian scleractinian corals. Marine and Freshwater Research 43(4):683-705.

Steward, J.S., R.W. Virnstein, L.J. Morris, and E.F. Lowe. 2005. Setting seagrass depth, coverage, and light targets for the Indian River Lagoon, Florida. Estuaries and Coasts 28:923-935.

Steward, J.S., R.W. Virnstein, M.A. Lasi, L.J. Morris, J.D. Miller, L.M. Hall, and W.A., Tweedale. 2006. The impacts of the 2004 hurricanes on hydrology, water quality, and seagrass in the central Indian River Lagoon, Florida. Estuaries and Coasts 29:954-965.

Storr, J.F. 1964. Ecology and oceanography of the coral-reef tract, Abaco Island Bahamas. Geol Sco Amer Spec Pap 79, 98 p.

Sutherland, K. P., and coauthors. 2010. Human sewage identified as likely source of white pox disease of the threatened Caribbean elkhorn coral, Acropora palmata. Environmental Microbiology 12(5):1122-1131.

Sutherland, K. P., J. W. Porter, and C. Torres. 2004. Disease and immunity in Caribbean and Indo-Pacific zooxanthellate corals. Marine Ecology Progress Series 266:273-302.

Szmant, A. M. 1986. Reproductive ecology of Caribbean reef corals. Coral Reefs 5(1):43-53.

Szmant, A. M. 2002. Nutrient enrichment on coral reefs: is it a major cause of coral reef decline? Estuaries and Coasts 25(4):743-766.

Szmant, A. M., and M. W. Miller. 2005. Settlement preferences and post-settlement mortality of laboratory cultured and settled larvae of the Caribbean hermatypic corals *Montastaea faveolata* and *Acropora palmata* in the Florida Keys, USA. Pages 43-49 *in* Proc. 10th Int Coral Reef Symposium.

Szmant, A. M., and N. J. Gassman. 1990. The effects of prolonged "bleaching" on the tissue biomass and reproduction of the reef coral *Montastrea annularis*. Coral Reefs 8(4):217-224.

Szmant, A. M., E. Weil, M. W. Miller, and D. E. Colón. 1997. Hybridization within the species complex of the scleractinan coral *Montastraea annularis*. Marine Biology 129(4):561-572.

Szmant, A.M. 1986. Reproductive ecology of Caribbean reef corals. Coral Reefs 5: 43 53.

Szmant-Froelich, A. 1985. The effect of colony size on the reproductive ability of the Caribbean coral *Montastrea annularis* (Ellis and Solander). Pages 295–300 *in* C. Gabrie, and B. Salvat, editors. 5th International Coral Reef Symposium, Tahiti.

Tarrant, A. M., M. J. Atkinson, and S. Atkinson. 2004. Effects of steroidal estrogens on coral growth and reproduction. Marine Ecology Progress Series 269:121-129.

Telesnicki, G. J., and W. M. Goldberg. 1995b. Comparison of turbidity measurement by nephelometry and transmissometry and its relevance to water quality standards. Bulletin of Marine Science 57(2):540-547.

Telesnicki, G., and W. Goldberg. 1995a. Effects of turbidity on the photosynthesis and respiration of two south Florida reef coral species. Bulletin of Marine Science 57:527-539.

Tomascik, T. 1990. Growth rates of two morphotypes of Montastrea annularis along a eutrophication gradient, Barbados, WI. Marine Pollution Bulletin 21(8):376-381.

Tomascik, T. and F. Sander. 1987. Effects of eutrophication on reef-building corals. I. Structure of scleractinian coral communities on fringing reefs, Barbados, West Indies. Marine Biology 94:53-75.

Tomascik, T., and F. Sander. 1987. Effects of eutrophication on reef-building corals. II. Structure of scleractinian coral communities on fringing reefs, Barbados, West Indies. Marine Biology 94(1):53-75.

Torquemada, Y.F., M.J. Durako, and J.L.S. Lizaso. 2005. Effects of salinity and possible interactions with temperature and pH on growth and photosynthesis of Halophila johnsonii Eiseman. Marine Biology 148:251-260.

Torres, J. L., and J. Morelock. 2002. Effect of terrigenous sediment influx on coral cover and linear extension rates of three Caribbean massive coral species. Caribbean Journal of Science 38(3-4):222-229.

Tunnell, J. W. J. 1988. Regional comparison of southwestern Gulf of Mexico to Caribbean Sea coral reefs. Pages 303-308 *in* Proceedings Of The Sixth International Coral Reef Symposium, Townsville, Australia.

Tunnicliffe, V. 1981. Breakage and propagation of the stony coral *Acropora cervicornis*. Proceedings of the National Academy of Science, 78:2427-2431.

Tunnicliffe, V. 1981. Breakage and propagation of the stony coral Acropora cervicornis. Proceedings of the National Academy of Sciences 78(4):2427-2431.

UNEP. 2010. Global coral disease database. Online at development.unep-wcmc.org/GIS/Coraldis/?CFID=5355509&CFTOKEN=41011411. Cambridge, UK.

Van Duyl, F. C. 1985. Atlas of the Living Reefs of Curacao and Bonaire (Netherlands Antilles). Vrije Universiteit, Amsterdam.

Van Moorsel, G. W. N. M. 1983. Reproductive strategies in two closely related stony corals (Agaricia, Scleractinia). Marine Ecology Progress Series 13:273-283.

van Tussenbroek, B.I. 1994. The impact of Hurricane Gilbert on the vegetative development of Thalassia testudinum in Puerto Morelos coral reef lagoon, Mexico: a retrospective study. Botánica Marina 37:421-428.

Van Veghell, M. and P. Hoetjes. 1995. Effects of tropical storm Bret on Curacao reefs. Bulletin of Marine Science, 56(2): 692-694.

Van Veghell, M. and P. Rolf. 1994. Reproductive characteristics of the polymorphic Caribbean reef building coral *Montastrea annularis*. III. Reproduction in damaged and regenerating colonies. Marine Ecology Progress Series, 109: 229-233.

Vardi, T. 2011. The threatened Atlantic elkhorn coral, *Acropora palmata*: population dynamics and their policy implications. dissertion. University of California, San Diego.

Vardi, T., D. E. Williams, and S. A. Sandin. 2012. Population dynamics of threatened elkhorn coral in the northern Florida Keys, USA. Endangered Species Research 19:157–169.

Vargas-Angel, B., J. D. Thomas, and S. M. Hoke. 2003. High-latitude *Acropora cervicornis* thickets off Fort Lauderdale, Florida, USA. Coral Reefs 22(4):465-473.

Vaughan, T. W. 1915. The geological significance of the growth rate of the Floridian and Bahamian shoal-water corals. Journal of the Washington Academy of Science 5:591-600.

Vaughan, T.W. 1915. The geological significance of the growth rate of the Floridian and Bahamian shoal-water corals. J. Wash. Acad. Sci., 5:591-600.

Vaughn, T. W. 1915. The geologic significance of the growth rate of the Floridian and Bahaman shoal water corals. Journal of the Washington Academy of Sciences 5:591-600.

Vermeij, M. J. A., K. L. Marhaver, C. M. Huijbers, I. Nagelkerken, and S. D. Simpson. 2010. Coral larvae move toward reef sounds. PLoS ONE 5(5):e10660.

Veron, J. E. N. 2000. Corals of the World. Australian Institute of Marine Science. Townsville, Australia 3 volumes.

Vijayavel, K., C. A. Downs, G. K. Ostrander, and R. H. Richmond. 2012. Oxidative DNA damage induced by iron chloride in the larvae of the lace coral Pocillopora damicornis. Comparative Biochemistry and Physiology C-Toxicology & Pharmacology 155(2):275-280.

Villinski, J. T. 2003. Depth-independent reproductive characteristics for the Caribbean reef-building coral *Montastraea faveolata*. Marine Biology 142(6):1043-1053.

Virnstein, R.W. and L.J. Morris. 2007. Distribution and abundance of Halophila johnsonii in the Indian River Lagoon: an update. SJRWMD Technical Memorandum #51. St. Johns River Water Management District, Palatka, Florida. 16 pp.

Virnstein, R.W. and L.M. Hall. 2009. Northern range extension of the seagrasses Halophila johnsonii and Halophila decipiens along the east coast of Florida, USA. Aquatic Botany 90:89-92.

Virnstein, R.W., L.C. Hayek, and L.J. Morris. 2009. Pulsating patches: a model for the spatial and temporal dynamics of the threatened seagrass Halophila johnsonii. Marine Ecology Progress Series 385:97-109.

Vollmer, S. V., and S. R. Palumbi. 2007. Restricted gene flow in the Caribbean staghorn coral *Acropora cervicornis*: Implications for the recovery of endangered reefs. Journal of Heredity 98(1):40-50.

Wagner, D. E., P. Kramer, and R. van Woesik. 2010. Species composition, habitat, and water quality influence coral bleaching in southern Florida. Marine Ecology Progress Series 408:65-78.

Wahle, C.M. 1983. Regeneration of injuries among Jamaican gorgonians: The roles of colony physiology and environment. Biology Bulletin, 164:778-790.

Walker, B.K., R.E. Dodge, D.S. Gilliam. 2008b. LIDAR-derived benthic habitat maps enable the quantification of potential dredging impacts to coral reef ecosystems. In proceeding of: ACES: A Conference on Ecosystem Services 2008: Using Science for Decision Making in Dynamic Systems, At Naples, Florida

Walker, B. K., E. A. Larson, A. L. Moulding, and D. S. Gilliam. 2012. Small-scale mapping of indeterminate arborescent acroporid coral (*Acropora cervicornis*) patches. Coral Reefs.

Wallace, C.C. 1985. Reproduction, recruitment and fragmentation in nine sympatric species of the coral genus *Acropora*. Marine Biology, 88:217-233.

Ward, J., and coauthors. 2006. Coral diversity and disease in Mexico. Diseases of Aquatic Organisms 69(1):23-31.

Ward, P., and M. J. Risk. 1977. Boring pattern of the sponge Cliona vermifera in the coral Montastrea annularis. Journal of Paleontology 51(3):520-526.

Warner, M. E., T. C. LaJeunesse, J. D. Robison, and R. M. Thur. 2006. The ecological distribution and comparative photobiology of symbiotic dinoflagellates from reef corals in Belize: potential implications for coral bleaching. Limnology and Oceanography 51(4):1887-1897.

Warner, M. E., W. K. Fitt, and G. W. Schmidt. 1996. The effects of elevated temperature on the photosynthetic efficiency of zooxanthellae in hospite from four different species of reef coral: a novel approach. Plant, Cell and Environment 19:291-299.

Weil, E., and N. Knowton. 1994. A multi-character analysis of the Caribbean coral Montastraea annularis (Ellis and Solander, 1786) and its two sibling species, M. faveolata (Ellis and Solander, 1786) and M. franksi (Gregory, 1895). Bulletin of Marine Science 55(1):151-175.

Weil, E., I. Urreiztieta, and J. Garzón-Ferreira. 2002. Geographic variability in the incidence of coral and octocoral diseases in the wider Caribbean. Proceedings of the 9th International Coral Reef Symposium 2:1231-1237.

Wells, J. W. 1933. A study of the reef Madreporaria of the Dry Tortugas and sediments of coral reefs. Cornell University, Ithaca, NY.

Wheaton, J. W., and W. C. Jaap. 1988. Corals and other prominent benthic cnidaria of Looe Key National Marine Sanctuary, FL.

Wheaton, J.W. and W.C. Jaap. 1988. Corals and other prominent benthic cnidaria of Looe Key National Marine Sanctuary, FL. Florida Marine Research Publication 43.

Whitfield, A. K. and M. N. Bruton (1989). "Some biological implications of reduced freshwater inflow into eastern Cape estuaries: a preliminary assessment." South African Journal of Science 85: 691-694.

Wilkinson, C., and D. Souter. 2008. Status of Caribbean coral reefs after bleaching and hurricanes in 2005, Townsville.

Williams, D. E., and M. W. Miller. 2005. Coral disease outbreak: pattern, prevalence and transmission in *Acropora cervicornis*. Marine Ecology Progress Series 301:119-128.

Williams, D. E., and M. W. Miller. 2012. Attributing mortality among drivers of population decline in Acropora palmata in the Florida Keys (USA). Coral Reefs 31(2):369-382.

Williams, D. E., M. W. Miller, and K. L. Kramer. 2008. Recruitment failure in Florida Keys *Acropora palmata*, a threatened Caribbean coral. Coral Reefs 27:697-705.

Williams, D. E., M. W. Miller, and K. L. Kramers. 2006. Demographic monitoring protocols for threatened Caribbean Acropora spp. corals.

Williams, E.H. and L. Bunkley-Williams. 1990. The world-wide coral reef bleaching cycle and related sources of coral mortality. Atoll Research Bulletin, 335:1-71.

Winkler, S. and A. Ceric. 2006. 2004 Status and trends in water quality at selected sites in the St. Johns River Water Management District. St. Johns River Water Management District, Technical Publication SJ2006-6, Palatka, Florida. 106 pp.

Woodley, J. D., and coauthors. 1981. Hurricane Allen's impact on Jamaican coral reefs. Science 214(4522):749-755.

Woodley, J.D., E.A. Chornesky, P.A. Clifford, J.B.C. Jackson, L.S. Kaufman, N. Knowlton, J.C.Lang, M.P. Pearson, J.W. Porter, M.C. Rooney, K.W. Rylaarsdam, V.J. Tunnicliffe, C.M. Wahle, J.L.Wulff, A.S.G. Curtis, M.D. Dallmeyer, B.D. Jupp, M.A.R. Koehl, J. Niegel, E.M Sides. 1981. Hurricane Allen’s impact on Jamaican coral reefs. Science, 214:749–755.

Woodward-Clyde Consultants. 1994. Biological resources of the Indian River Lagoon. Final Technical Report. Prepared for the Indian River Lagoon National Estuary Program, July 1994.

Wooldridge, S. A. 2009. Water quality and coral bleaching thresholds: Formalising the linkage for the inshore reefs of the Great Barrier Reef, Australia. Marine Pollution Bulletin 58(5):745-751.

Yap, H.T. and E.D. Gomez. 1984. Growth of *Acropora pulchra*. II. Responses of natural and transplanted colonies to temperature and day length. Marine Biology, 81:209-215.

Zimmer, B., W. Precht, E. Hickerson, and J. Sinclair. 2006. Discovery of *Acropora palmata* at the Flower Garden Banks National Marine Sanctuary, northwestern Gulf of Mexico. Coral Reefs 25:192.

Zubillaga, A. L., C. Bastidas, and A. Cróquer. 2005. High densities of the Elkhorn coral Acropora palmata in Cayo de Agua, Archipelago Los Roques National Park, Venezuela. Coral Reefs 24(1):86.

Zubillaga, A. L., L. M. Marquez, A. Croquer, and C. Bastidas. 2008. Ecological and genetic data indicate recovery of the endangered coral Acropora palmata in Los Roques, Southern Caribbean. Coral Reefs 27(1):63-72.

APPENDIX A

Background

Vessel Strike Avoidance Measures and Injured or Dead Protected Species Reporting
NOAA Fisheries Service, Southeast Region

NOAA Fisheries Service has determined that collisions with vessels can injure or kill protected species (e.g., endangered and threatened species, and marine mammals). The following standard measures are recommended to reduce the risk associated with vessel strikes or disturbance of these protected species. NOAA Fisheries Service should be contacted to identify any additional conservation and recovery issues of concern for protected species in your operating area.

Protected Species Identification Training

Vessel crews should use an Atlantic and Gulf of Mexico reference guide that helps identify the species of marine mammals and sea turtles that might be encountered in U.S. waters of the Atlantic Ocean, including the Caribbean and Gulf of Mexico. Additional training should be provided regarding information and resources available regarding federal laws and regulations for protected species, ship strike information, critical habitat, migratory routes and seasonal abundance, and recent sightings of protected species.

Vessel Strike Avoidance

The following measures must be taken in order to avoid causing injury or death to marine mammals and sea turtles:

1. Vessel operators and crews will maintain a vigilant watch for marine mammals and sea turtles to avoid striking sighted protected species.

2. When whales are sighted, maintain a distance of 100 yards or greater between the whale and the vessel.

3. When sea turtles or small cetaceans are sighted, attempt to maintain a distance of 50 yards or greater between the animal and the vessel whenever possible.

4. When small cetaceans are sighted while a vessel is underway (e.g., bow-riding), attempt to remain parallel to the animal's course. Avoid excessive speed or abrupt changes in direction until the cetacean has left the area.

5. Reduce vessel speed to 10 knots or less when mother/calf pairs, groups, or large assemblages of cetaceans are observed near an underway vessel, when safety permits. A single cetacean at the surface may indicate the presence of submerged animals in the vicinity; therefore, prudent precautionary measures should always be exercised. The vessel will attempt to route around the animals, maintaining a minimum distance of 100 yards whenever possible.

6. Whales may surface in unpredictable locations or approach slowly moving vessels. When an animal is sighted in the vessel's path or in close proximity to a moving vessel, reduce speed and shift the engine to neutral. Do not engage the engines until the animals are clear of the area.

Additional Requirements for the North Atlantic Right Whale

1. If a sighted whale is believed to be a North Atlantic right whale, federal regulation requires a minimum distance of 500 yards be maintained from the animal (50 CFR 224.103 (c)).

2. Vessels entering North Atlantic right whale critical habitat are required to report into the Mandatory Ship Reporting System.

3. Mariners should check with various communication media for general information regarding avoiding ship strikes and specific information regarding North Atlantic right whale sighting locations. These include NOAA weather radio, U.S. Coast Guard NAVTEX broadcasts, and Notices to Mariners.

Injured or Dead Protected Species Reporting

Vessel crews will report sightings of any injured or dead protected species immediately, regardless of whether the injury or death is caused by your vessel.

Report marine mammals to the Southeast U.S. Stranding Hotline: 305-862-2850
Report sea turtles to the Southeast Regional Office: 727-824-5312

If your vessel is responsible for the injury or death, the responsible parties will remain available to assist the respective salvage and stranding network as needed. In addition, if the injury or death was caused by a collision with your vessel, you must notify the Southeast Regional Office immediately of the strike by telephone at (727) 824-5312, or by fax at (727) 824-5309. The report should include the following information:

a. the time, date, and location (latitude/longitude) of the incident;

b. the name and type of the vessel involved;

c. the vessel's speed during the incident;

d. a description of the incident;

e. water depth;

f. environmental conditions (e.g., wind speed and direction, sea state, cloud cover, and visibility);

g. the species identification or description of the animal, if possible; and
h. the fate of the animal.

For additional information, please contact the Protected Resources Division at:

NOAA Fisheries Service
Southeast Regional Office
263 13th Avenue South
St. Petersburg, FL 33701

Tel: (727) 824-5312
Visit us on the web at http://sero.nmfs.noaa.gov

APPENDIX B

Transplantation Protocols for Port Everglades Expansion Project.

All relocation field activities, data collection, analysis and reporting will be supervised by a marine biologist (minimum academic requirement is M.S. degree in related field, or equivalent experience) with experience in coral transplantation and survival monitoring. The qualifications of any persons conducting transplantation work must be submitted to NMFS Protected Resources Division, for review.

The colonies will be collected carefully using a hammer and chisel. Upon collection, the colonies must be kept in bins and maintained in seawater at all times. During transportation to the transplant site, the corals must be covered. Transplantation should occur as soon as operationally feasible, and no more than 24 hours after the colony is removed from its original location. The collected colonies must be kept at the original depth until transplantation commences (i.e., cached on site).

The USACE must ensure that all transplanted colonies are re-located to suitable habitat near their original location. The colonies must be transplanted no closer than 400 feet (ft) from the project area (550 ft from the edge of channel) in an area of suitable habitat/substrate resembling that of the colonies original location as soon as operationally feasible. For the purposes of this opinion, suitable habitat is considered: similar depth as origin (+/- 5 ft); means consolidated hardbottom (to include the artificial boulder reef site) or dead coral skeleton that is free from fleshy macroalgae cover and sediment cover occurring in water depths from the mean high water (MHW) line to 30 meters (98 ft); appropriate water quality (based on water quality data and local knowledge), and minimal chances of other disturbances (boat groundings, damage caused by curious divers/fisherman). All efforts should be made to transplant the fragment to the same depth from which it was removed (i.e., +/- 5 ft).

The material used to attach the colonies to suitable substrate must be Portland cement. Before applying the Portland cement to the substrate, it must be cleaned of any sediment or algae. The Portland cement should then be taken out of the dry lock bag and pressed against the clean substrate. The transplanted colonies must then be pressed gently into the Portland cement with proper care. Transplanted colonies must be no closer than 0.75 meters from one another.

To assist in monitoring efforts, a plastic identification tag must be attached adjacent to each transplanted colony. Finally, the collected location, length, width, depth and orientation of each colony to be transplanted will be recorded. The transplanted location and depth of each colony, as well as the species and identification number, will be recorded.

APPENDIX C

Calculations for Port Everglades mitigation

Area of impact = 21.66 acres
From the draft *Acropora* Recovery Plan, density of 1 colony (≥0.5 m diameter) per m^2 in 5% of consolidated habitat 5-20 m depth.
5% of 21.66 acres = 1.083 acres = 4,382.7 m^2
<u>Area of a square</u>: Colony 0.5*0.5 m = 0.25 m^2 coral occupancy per m^2 of hardbottom
Assume outplanted colonies will be 0.2 m diameter: 0.2*0.2 m=0.04 m^2 in area
0.25m^2/0.04m^2=6.25 colonies needed per m^2 hardbottom
4,382.7 m^2 * 6.25 colonies ≈ **27,392 colonies needed**
<u>Area of a circle</u>: colony 0.5 m diameter = 0.2 m^2 coral occupancy per m^2 of hardbottom
Assume outplanted colonies will be 0.2 m diameter: 0.1^2 * 3.14 = 0.03 m^2 in area
0.2 m^2/0.03 m^2=6.67 colonies needed per m^2 hardbottom
4,382.7 m^2 * 6.67 colonies ≈ **29,233 colonies needed**

Based on data from Gilliam (see table below)
Assume colony mortality of 44% per year
Assume partial mortality rate of 25% per year (75% tissue survival)
Assume colony growth rate of 8 cm per year (diameter)
Assume recruitment rate of 22% per year

Per m^2 hardbottom:
colonies: 6.25-2.75+1.4=4.9 colonies
coral tissue growth: 4.9 colonies* (0.28 m*0.28 m)=0.38 m^2
tissue survival: 0.75*0.38 m^2= 0.28 m^2

Conclusion: With the very rough estimates of recruitment, mortality, and growth rates, an estimated 27,392 to 29,233 colonies will need to be outplanted to maintain the density/coverage criteria from the Acropora Recovery Plan.

Notes for Port Everglades expansion mitigation

Brian Walker (Pers. Comm.) NSU/NCRI

Broward County	m^2	km^2	acres	hectares
Coral Reef and Colonized Hardbottom	44,689,106	44.68911	11,042.92	4,468.911
Known *A. cervicornis* area	155,000	0.155	38.30133	15.5

K. Wirt (Pers. Comm.) FL FWC/USF

Approximately 600 *A. cervicornis* sightings in Broward (see map)

(Vargas-Angel et al. 2003)

Thickets: 1,000 m^2 to ~8,000 m^2 in area

% cover: ~5-28%

Recruit (< 5 cm dia) density: 0-1 m^{-2}; mean 0.1 m^{-2}

Largest colony diameter: 1.8 m

Largest colony density: 3 m^{-2}

Mean cover affected by WBD: 1.8%

(Walker et al. 2012)

Thickets: ~10,400 m^2 and 22,500 m^2

(Hollarsmith et al. 2012)

Outplanted 1 yr old colonies 20-40 cm diameter

Data from D. Gilliam, NSU

Nova Southeastern University Oceanographic Center											
David S. Gilliam											
15-Nov-2013											
***Acropora cervicornis* data**											
Project	**Duration of Project**	**n_i**	**n_e**	**% Survival**	**Mean change in whole colony diameter (cm) ± SD**	**Mean % Live**					

Broward *Acropora* mapping project	2 yrs	63	33	52%	10.5 ± 22	71 ± 29					
2007 Nursery Donor Colonies	1.5 yrs	10	6	60%	8 ± 31	66 ± 39					
2010 Nursery Donor Colonies	2 yrs	20	11	55%	5.5 ± 22	87 ± 19					
Recruitment											
Site 1-Broward	Fall 2010	Winter 2011	Summer 2011	Fall 2011	Winter 2012	Summer 2012	Fall 2012	Winter 2013	Summer 2013	Fall 2013	
Colonies	271	392	548	405	599	681	390	412	419	501	
Fragments	514	364	410	526	462	249	506	394	315	403	
Site 2-Broward											
Colonies	122	125	163	159	201	233	147	202	202	203	
Fragments	111	48	87	235	266	108	190	129	98	139	
<u>Notes</u>											
Survivorship											
n_i= number of colonies at beginning of project											
n_e= number of colonies remaining at end of project monitoring											
Colonies were considered "dead" if they went missing. A majority of the colonies that we lost were due to colony dislodgement, this does not necessarily mean they are dead, but may have fragmented and attached elsewhere at the site.											
Growth											

Only whole colony size was measured for these projects. Change in colony size came from the change in final colony max diameter from the initial colony max diameter over the length of the project											
Recruitm ent											
Colonies and fragments are counted 3 times a year within permanent monitoring stations. These are not fate tracked but may give an indication of recruitment through asexual reproduction.											

Map from K. Wirt (FWC)